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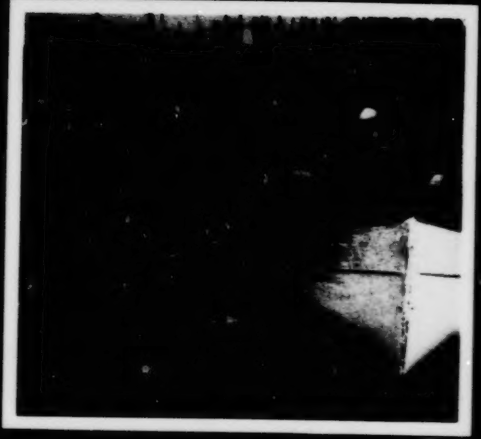

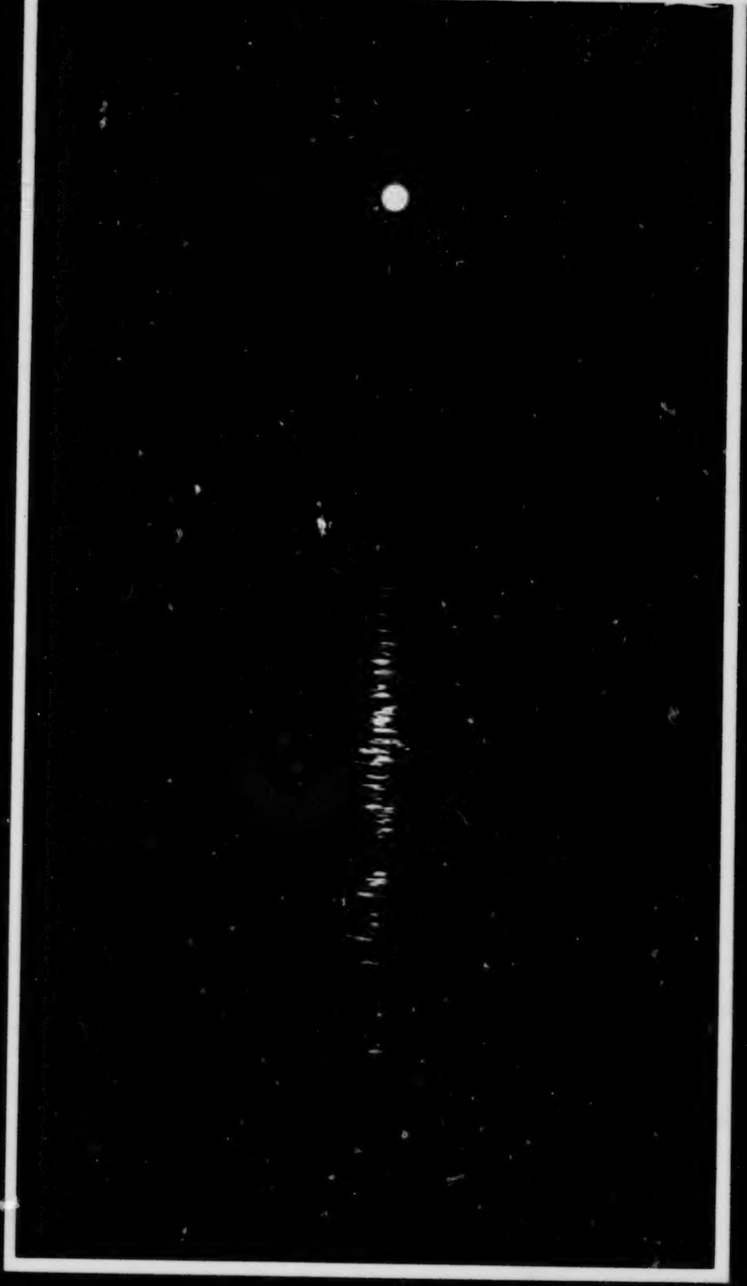

SUMMARY REPORT

Canada - Manitoba Agreement on the Study and Monitoring of Mercury in the Churchill River Diversion

SUMMARY REPORT

Canada - Manitoba Agreement on the Study and Monitoring of Mercury in the Churchill River Diversion

Canada
Manitoba

Canada-Manitoba Mercury Agreement

March 23, 1987

The Honourable Gerard Lecuyer
Minister
Environment and Workplace Safety
and Health
Room 156, Legislative Building
WINNIPEG, Manitoba

The Honourable Tom McMillan
Minister
Environment Canada
28th Floor, Les Terrasses de la Chaudiere
HULL, Quebec

Gentlemen:

We are pleased to submit the Summary Report and Technical Appendices of the Canada-Manitoba Agreement on the Study and Monitoring of Mercury in the Churchill River Diversion.

It has been our pleasure to have worked with the many individuals who have been involved in this Agreement. Their dedication to the Agreement's objectives, their pursuit of scientific excellence, and their spirit of cooperative effort have made our job much easier.

It is our hope that the Summary Report has succeeded in presenting the highlights of the many Agreement studies in an objective and comprehensible manner.

Respectfully submitted,

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EXECUTIVE SUMMARY

The Lake Winnipeg, Churchill-Nelson River Hydroelectric Development Project is located in northern Manitoba, Canada. One component of this large scale, complex project is the diversion of approximately 75% of the flow of the Churchill River into the Nelson River. This diversion, which became fully operational in 1977, has resulted in extensive areas of flooded terrain and eroding shorelines along the Churchill River Diversion route.

Elevated mercury levels in fish from the lakes along the Churchill River Diversion route were first detected in 1978/79 as part of the routine monitoring of commercial fish shipments. Concern about the possible impact of those elevated mercury levels on the health and economic livelihood of the local people led to a federal-provincial agreement to study the mercury problem.

There is no evidence that man-made sources of mercury are an important source of mercury in the Agreement area. The natural levels of mercury found in the soil, water, and sediment of the Agreement area are not elevated in comparison to the reported levels from other non-polluted areas. Yet the mercury levels in fish along the diversion route are elevated in comparison to historical levels, to the mercury levels in the fish from other northern Manitoba lakes, and to the Canadian guidelines for mercury levels in fish for human consumption.

Mercury levels in the people of South Indian Lake and Nelson House (the two native communities located along the diversion route) are generally within the normal range of 0-20 parts per billion in blood. However, the occurrence of mercury levels above the normal range in some women of child-bearing age in these communities is cause for concern, because of the sensitivity of the fetus to mercury poisoning.

Mercury levels in mink and otter,which are fish eating mammals, from the Agreement area are elevated in comparison to other parts of Manitoba, but are below toxic levels.

Methyl mercury, the predominant form of mercury found in fish, is produced naturally in aquatic environments in small amounts. It is produced (methylated) and destroyed (demethylated) by bacteria. The net production of methyl mercury is determined by the relative rates of methylation and demethylation.

Agreement studies found that the net methylation rates were highest in areas of flooded terrain. Studies also showed a strong positive correlation between net mercury methylation rates and mercury levels in fish. These two findings explain why the lakes in the Agreement area which were the most extensively flooded contained fish with the highest mercury concentrations.

Agreement studies have shown that the most important factor affecting net mercury methylation rates is the amount and type of flooded organic material. It was found, at least in the case of moss-peat material, that a doubling of the amount of organic material resulted in a doubling of the amount of mercury which accumulated in the fish. It was also found that a wide variety of organic materials could simulate the bioaccumulation of mercury in fish, although they differed in their abilities to do so. This indicates that the effect of flooding on mercury levels in fish is a general response.

Another important factor affecting net mercury methylation rates is the amount of oxygen present. It was found that the net production of methyl mercury was higher when oxygen levels were very low.

There is no indication that net mercury methylation rates are declining in the flooded areas of the lakes along the Churchill River Diversion route. It is estimated that it will take decades for the mercury levels in the predatory fish from these lakes to return to historical (pre-impoundment) levels. The mercury levels in the whitefish from Southern Indian Lake have declined from their recorded post-impoundment peak.

The results of the Agreement and other recent studies indicate that elevated mercury levels in fish following impoundment is a widespread problem. Elevated mercury levels must be considered explicitly in the planning and costing of all future impoundment/reservoir projects.

RÉSUMÉ

Le projet d'aménagement hydroélectrique lac Winnipeg-Churchill-Nelson est en cours de réalisation dans le nord du Manitoba, au Canada. Dans le cadre de ce project complexe, de grande envergure, environ 75 p. 100 du cours du Churchill est dirigé vers le Nelson; cette vaste enterprise a été achevée en 1977. D'immenses étendues de terre ont donc été inondées et les berges du nouveau cours du Churchill ont subi les attaques de l'érosion.

On a détecté pour la première fois en 1978-1979 des concentrations élevées en mercure dans les poissons des lacs le long du nouveau cours du Churchill en procédant aux contrôles d'usage des prises commerciales expédiées. Cette situation était de nature à menacer la santé et le bien-être économique des populations de l'endroit et, pour cette raison, une entente fédérale-provinciale a été conclue en vue d'étudier ce problème.

Rien ne permet d'établir que les sources artificielles de mercure représentent un apport de mercure important dans le territoire visé par l'Entente. Les niveaux naturels de mercure que l'on trouve dans le sol, l'eau et les sédiments du territoire visé par l'Entente ne sont pas élevés par rapport à ceux qui sont signalés dan d'autres entroits non polluées. Par contre, les concentrations en mercure dans les poissons pris le long du nouveau cours du Churchill sont élevés comparativement aux niveaux antérieurs, aux concentrations dans les poissons d'autres lacs du nord du Manitoba ainsi qu'aux valeurs précisées dans les directives canadiennes sur la concentration en mercure dans les poissons destinés à la consommation humaine.

De façon générale, les concentrations en mercure chez les gens de South Indian Lake et de Nelson House (les deux collectivités amérindiennes le long du nouveau cours du Churchill) se situent dans la limite de 0 à 20 parties par milliard dans le sang. Toutefois on a décelé dans cex localités des concentrations en mercure au-dessus de la normale chez certaines femmes en âge procréer, ce qui est inquiétant puisque le foetus est trés sensible à l'empoisonnenment par le mercure.

Les concentrations en mercure chez le vision et la loutre - mammifères qui se nourrissent de poissons - dans le territoire visé par l'Entente sont élevées comparativement à d'autres endroits du Manitoba, mais elles ne dépassent toutefois pas les niveaux de toxicité.

Le mercure-méthyle, forme sous laquelle on retrouve surtout le mercure dans les poissons, est produit naturellement en petites quantités dans les milieux aquatiques. Le mercure sous cette forme est produit (méthylé) et détruit (déméthylé) par des bactéries; la production nette de mercure-méthyle est déterminée par les taux relatifs de méthylation et de déméthylation.

Les études aux termes de l'Entente ont démontré que les taux nets de méthylation étaient les plus élevés dans les terrains inondés; elles ont également démontrée une forte corrélation positive entre les taux nets de méthylation du mercure et les niveaux de mercure dans les poissons. Ces deux conclusions expliquent pourquoi, dans le territoire visé par l'Entente, les lacs les plus touchés par l'inondation sont ceux où on trouvait le poisson ayant les plus fortes concentrations en mercure.

Les études aux termes de l'Entente ont démontré que le facteur l'influence est déterminante sur les taux nets de méthylation du mercure reste la quantité et le type de matière organique inondée. On a établi, au moins pour ce qui est de la tourbe mousseuse, qu'une quantité deux fois plus grande de matière organique se traduit par une concentration de mercure deux fois plus élevée dans les poissons. Il a également été établi qu'une grande variété de matières organiques pouvait stimuler la bioaccumulation de mercure chez le poisson, bien qu'elles agissent à des degrés différents. L'effet de l'inondation sur les concentrations en mercure dans les poissons constitue donc une réaction générale.

La quantité d'oxygène présente est un autre facteur important qui influe sur les taux nets de méthylation du mercure; dans ce cas, il a été établi que la production nette de mercure-méthyle est plus élevée lorsque les niveaux d'oxygène sont trés bas.

Rien ne permet de dire que les taux nets de méthylation du mercure sont en régression dans les secteurs inondés des lacs le long du nouveau cours du Churchill. On estime qu'il faudra des décennies pour que les concentrations en mercure dans les poissons prédateurs de ces lacs reviennent à leur niveau d'antan (avant la création du bassin de retenue). Les niveaux de mercure dans le corégone du lac Southern Indian ont fléchi par rapport aux sommets enregistrés après la création du bassin de retenue.

Les résultats de l'Entente et d'autres études effectuées récemment indiquent que les niveaux de mercure élevés dans les poissons après la création d'un bassin de retenue constituent un problème généralisé; il faut tenir compte très explicitement des niveaux élevés de mercure dans la planification et l'établissement des coûts relatifs à tout projet futur de création de réservoir ou de bassin de retenue.

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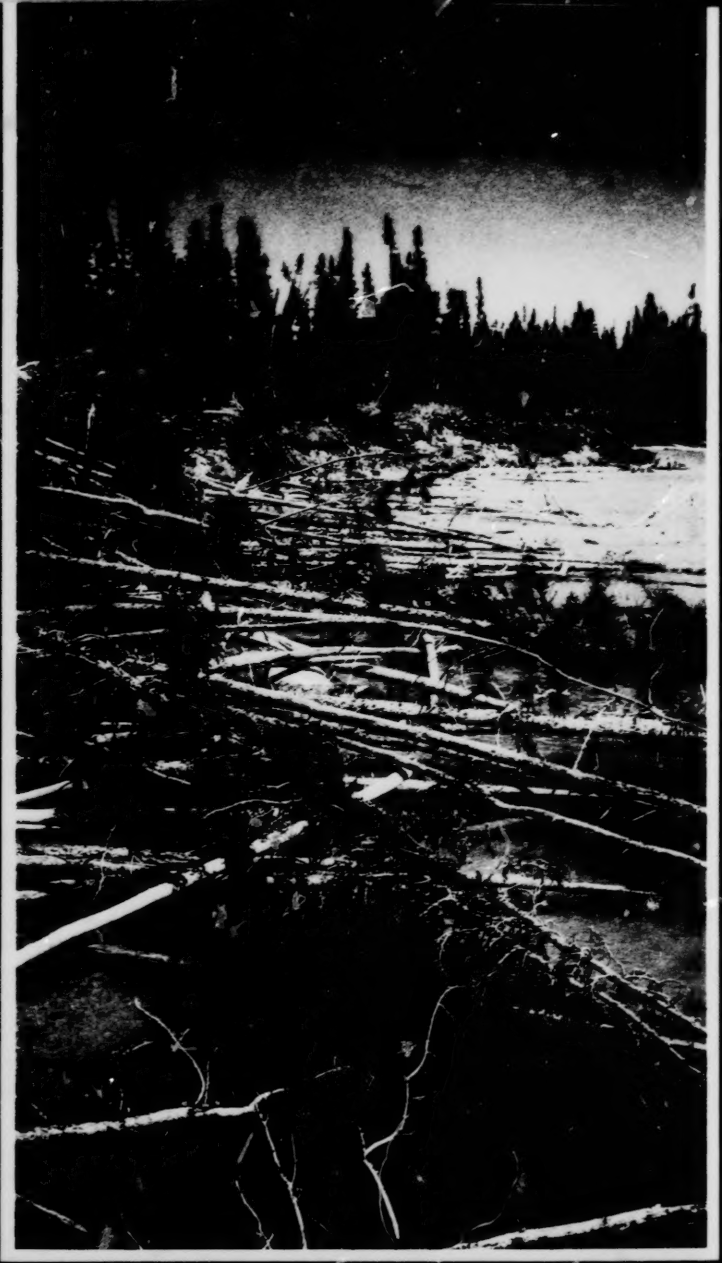
A NOTE ON UNITS

The units commonly used to report mercury concentration data vary with the component of the environment. In order to assist the reader in understanding the various units used in this report, the following definitions are provided:

ppm = parts per million
ppb = parts per billion
1 ppm = 1000 (or 10^3) ppb.
1 g = 1 gram
1 ug = 1 microgram = 10^{-6} grams
1 ng = 1 nanogram = 10^{-9} grams
ug·g = micrograms/gram
ug·L^{-1} = micrograms/litre
1 ug·g^{-1} = 1 ppm
1 ug·L^{-1} = 1 ppb
1 ug·Kg^{-1} = 1 ppb

Other units used in this report are:

1 Km = 1 kilometre = 0.62 mile
m^3.s^{-1} = cubic metres per second
= 35.31 cubic feet per second
1 ha = 1 hectare = 2.47 acres



CHAPTER 1

INTRODUCTION

- THE STUDY AREA
- THE AGREEMENT
- SOURCES OF MERCURY

Through the northern part of the province of Manitoba flow two of the largest rivers in western Canada, the Churchill and Nelson Rivers. The potential for hydroelectric development of these rivers has been recognized since the beginning of the twentieth century. Following feasibility studies conducted in the 1960's, the decision to proceed with a major hydroelectric project was announced in 1966.

The Churchill-Nelson Rivers Hydroelectric Development Project has resulted in a variety of significant environmental impacts. Some of these impacts were predicted, at least in kind, if not by degree. One significant impact that was not predicted was an increase in the **Mercury*** levels in the fish along the diversion route.

The elevated mercury levels in the fish of the diversion route lakes raised concerns about the potential effects on the health and economic livelihood of the local Cree population. These concerns led to the signing of the Canada-Manitoba Agreement on the Study and Monitoring of Mercury in the Churchill River Diversion. The research and monitoring programs undertaken under this Agreement to increase our knowledge and understanding of mercury along the diversion route are the subjects of this report.

Chapter 1 describes the area of study, with particular emphasis on the environmental changes that occurred as a result of the hydroelectric project. Also included in this chapter are a summary of the Agreement's salient features and a discussion of the source of mercury in this area.

Chapter 2 describes the mercury levels in the fish of the study area. Chapter 3 presents the findings of the research on the mechanisms which led to the elevated mercury levels in fish. Chapter 4 focuses on the mercury levels in the people of the study area, with a concluding section on the mercury levels in aquatic mammals. The duration and extent of the mercury problem is the subject of Chapter 5. Chapter 6 presents the conclusions and recommendations of the Agreement's studies.

STUDY AREA

Characteristics

The Churchill River flows for over 1600 kilometers from its origin in the Province of Alberta to its mouth at Hudson Bay. It had a pre-development discharge at Missi Falls (the outlet of Southern Indian Lake) of approximately 1011 cubic metres per second ($m^3.s^{-1}$). (A map of the Agreement area is located at the back of this report.)

The Nelson River flows 660 kilometers from its source at the outlet of Lake Winnipeg to Hudson Bay, descending some 217 metres of which approximately 152 metres is downstream of Split Lake. The average flow of the Nelson River at its source is approximately 1960 $m^3.s^{-1}$

The Nelson River parallels the lower Churchill River (from Southern Indian Lake to Hudson Bay) and is some 160 kilometers closer to the power demand center in southern Manitoba. Located between these two rivers, in the vicinity of the southern end of Southern Indian Lake, are the Rat and Burntwood Rivers. It was through these rivers that a major portion of the flow of the Churchill River was diverted into the Nelson River. It is this area, referred to in this report as the Churchill River diversion route, which was the main focus of the Agreement studies. Other northern Manitoba lakes were also studied, although less intensively, for purposes of comparison and controls for laboratory and field experiments.

The diversion route is located within the Precambrian Shield, in an area of **Glacio-Lacustrine** sedimentation (Fig. 1.1). The climate of the region is continental, consisting of long cold winters and short cool summers. The boreal forest vegetation includes black spruce, tamarack and jackpine as well as areas of deciduous species such as aspen, balsam poplar and willow. Sphagnum moss peat deposits are also common throughout the area. **Permafrost** is widespread throughout the area.

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* Technical words which are defined in the glossary are capitalized on their first appearance in the text.

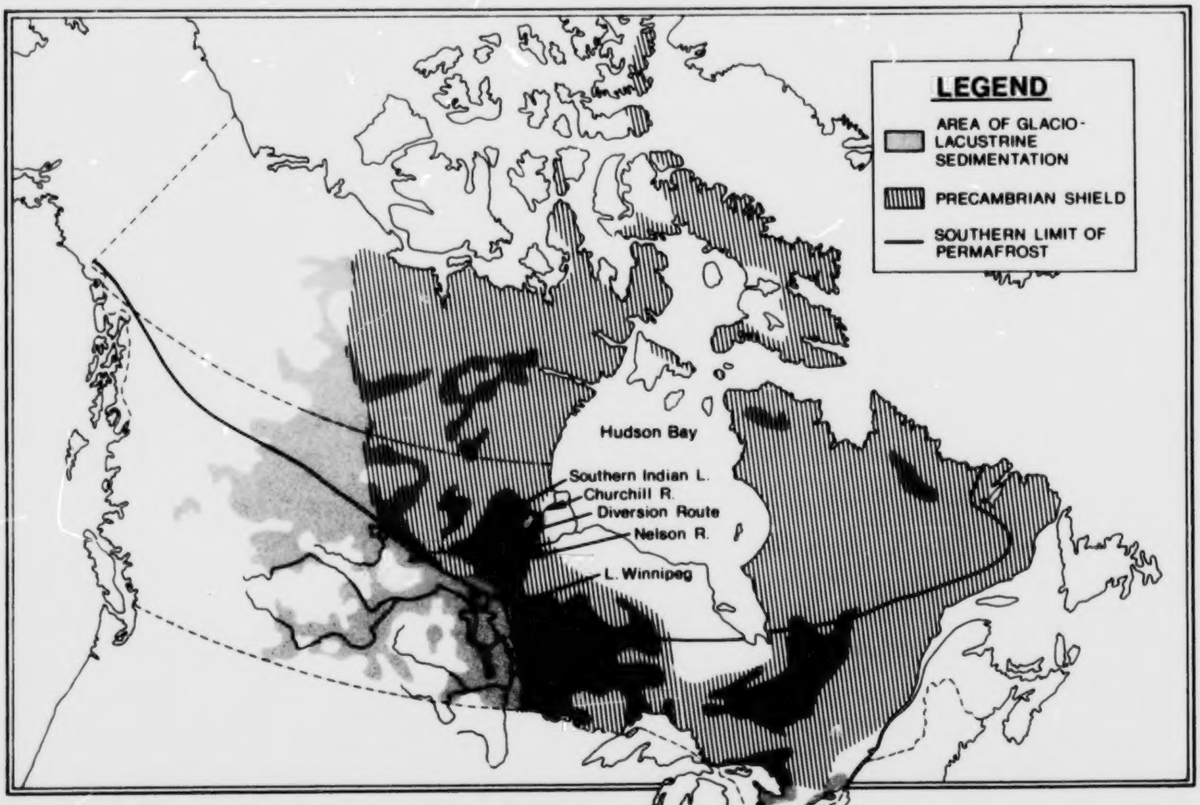
Two native communities, South Indian Lake and Nelson House, are located along the diversion route. Nelson House, on the shores of Footprint Lake, is an Indian reserve with a population of approximately 1,300. South Indian Lake, located on the east side of Southern Indian Lake near South Bay, has a population of approximately 700. The residents of both communities have strong economic and cultural ties with their local environments. Traditional pursuits of fishing, trapping, and hunting continue to be an important part of their life-styles. The City of Thompson, located further downstream on the diversion route, is a mining community with a population of approximately 15,000.

Hydroelectric Project

The Churchill River Diversion is part of a large and complex hydroelectric development project with a total design capacity of approximately 8400 MW. This project also involves the regulation of Lake Winnipeg and the construction and operation of fourteen generating stations.

Diversion of the Churchill River was accomplished by placing a control dam at the natural outlet of Southern Indian Lake (Missi Falls) in 1976 and raising the level of the lake approximately 3 metres above the long term mean. Water from Southern Indian Lake now leaves through a channel constructed between South Bay and the headwaters of the Rat River. A control structure at the outlet of Notigi Lake regulates flows down the Rat River valley. The Notigi Control Structure was closed in 1974, retaining local runoff water until water levels in the Rat River valley and Southern Indian Lake

Fig. 1.1 **Geographical Setting of the Churchill River Diversion Area**



(Source: Canadian Journal of Fisheries and Aquatic Sciences, Vol. 41, No. 4, 1984) were similar. The diversion channel was then opened in 1976 and the diversion was operating at full design capacity by late 1977.

To summarize, the diversion route begins at Southern Indian Lake, continues south through the diversion channel, down the Rat and Notigi Lakes, through the Notigi Control Structure and into Wapisu, Threepoint and Wuskwatim Lakes. It then continues along the Burntwood River, past Thompson, eventually meeting the Nelson River at Split Lake.

The terms of Manitoba Hydro's operating licence specify that the flow down the diversion route must not exceed 850 $m^3.s^{-1}$. Minimum releases must also be maintained at Missi Falls. The net result is that approximately 75% of the Churchill River was diverted to flow southward into the lower Nelson River valley. The combined annual flow of the Churchill and Nelson Rivers on the lower Nelson River, where most of the hydroelectric power is to be produced, is about 3500 $m^3.s^{-1}$.

The completion and operation of the diversion resulted in a significant alteration of the pre-existing lake levels and stream flows as illustrated in Figs. 1.2, 1.3, and 1.4. Flows, before and after diversion, are shown in Fig. 1.2 by a schematic diagram of the river systems. Fig. 1.3 illustrates the changes in selected stream flows and lake levels immediately before and after diversion. A perspective of the size of the diversion is provided in Fig. 1.4 by a comparison to the stream flows of a number of rivers.

Environmental Changes

The environmental changes which occurred along the diversion route varied considerably, most especially between Southern Indian Lake and Notigi Reservoir.

Southern Indian Lake, which experienced a raise in lake level of approximately 3 meters increased 21% in total lake area, from 1977 to 2391 km^2. Although all regions had the same increase in lake level, the relative increase in flooded area varied from region to region because of differences in local topography.

Raising the lake level caused a marked change in the type of materials at the eroding face of the shoreline. After flooding the amount of bedrock-controlled shoreline decreased from 88% to 14% and the amount of permafrost affected, glacio-lacustrine clay shorelines increased. Consequently, the amount of shoreline erosion was extensive and high concentrations of sediment were suspended in the lake.

Flows from the Churchill River diversion and the retention of local runoff have increased the surface elevations of most lakes in the Notigi forebay area by approximately 10 metres. The greatest water level increase (greater than 15 metres) occurred on Notigi Lake immediately above the Notigi Control Structure, resulting in an increase of 435% in the surface area of this lake (Fig. 1.5).

In contrast to Southern Indian Lake, Notigi Reservoir experienced little shoreline erosion and therefore low concentrations of suspended sediment. However scenes of submerged and dying vegetation are extensive here. Notigi Reservoir followed a more classical reservoir response than did Southern Indian Lake, with a period of increased biological activity and growth in all levels of the food chain. Widespread oxygen depletion occurred in Notigi Reservoir in 1974 and 1975. However, with the commencement of diversion flows in 1976, oxygen levels improved in all but the backwater areas.

Flooding along the lower portion of the Churchill River diversion route has not been as severe as it has in the Notigi Reservoir. Water levels downstream of the Notigi Control Structure have increased 3 to 5 metres above the historical mean elevations. Increased water levels and river flows have caused significant shoreline erosion and high turbidities.

The above mentioned differences in the environmental impacts in Southern Indian Lake and Notigi Reservoir proved significant in terms of their ultimate effect on mercury levels in fish. The hydroelectric development project and its environmental impact have been examined in great detail and reported on in a number of publications. Two particularly excellent references are the Lake Winnipeg, Churchill-Nelson Study Board Report (1975), and the Canadian Journal of Fisheries and Aquatic Sciences, Vol. 41, No. 4, 1984.

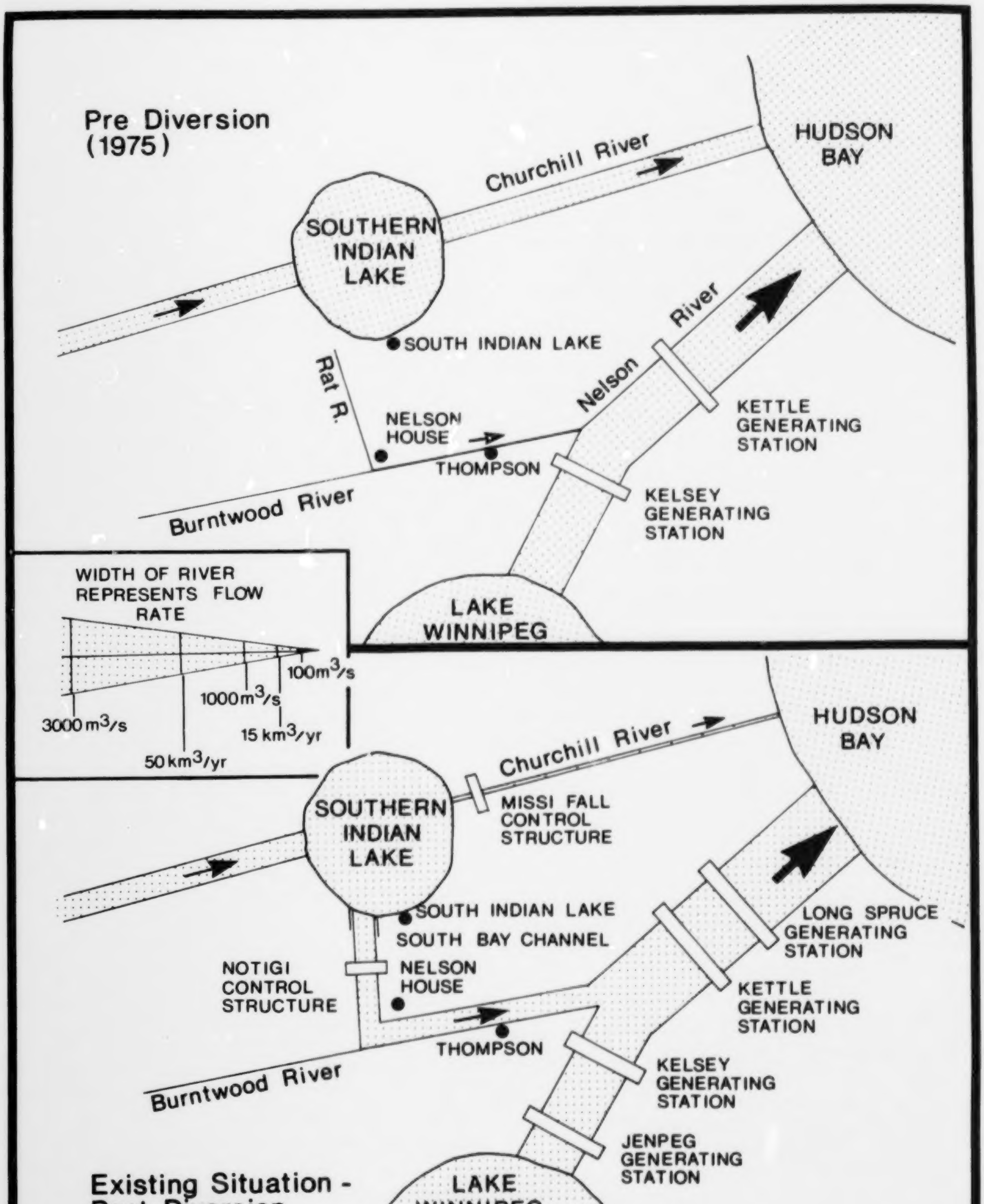

Pre Diversion
(1975)
HUDSON BAY
Churchill River
SOUTHERN INDIAN LAKE
SOUTH INDIAN LAKE
Nelson River
Rat R.
NELSON HOUSE
THOMPSON
KETTLE GENERATING STATION
KELSEY GENERATING STATION
Burntwood River
LAKE WINNIPEG
WIDTH OF RIVER REPRESENTS FLOW RATE
100m3/s
1000m3/s
3000 m3/s
15 km3/yr
50 km3/yr
HUDSON BAY
Churchill River
SOUTHERN INDIAN LAKE
MISSI FALL CONTROL STRUCTURE
SOUTH INDIAN LAKE
SOUTH BAY CHANNEL
LONG SPRUCE GENERATING STATION
NOTIGI CONTROL STRUCTURE
NELSON HOUSE
THOMPSON
KETTLE GENERATING STATION
KELSEY GENERATING STATION
JENPEG GENERATING STATION
Burntwood River
LAKE WINNIPEG
Existing Situation - Post Diversion

Fig. 1.2 **Pre- and Post-Diversion River Flows**

Fig. 1.3 **River Flows and Water Levels for Selected Sites, 1972 - 1982**

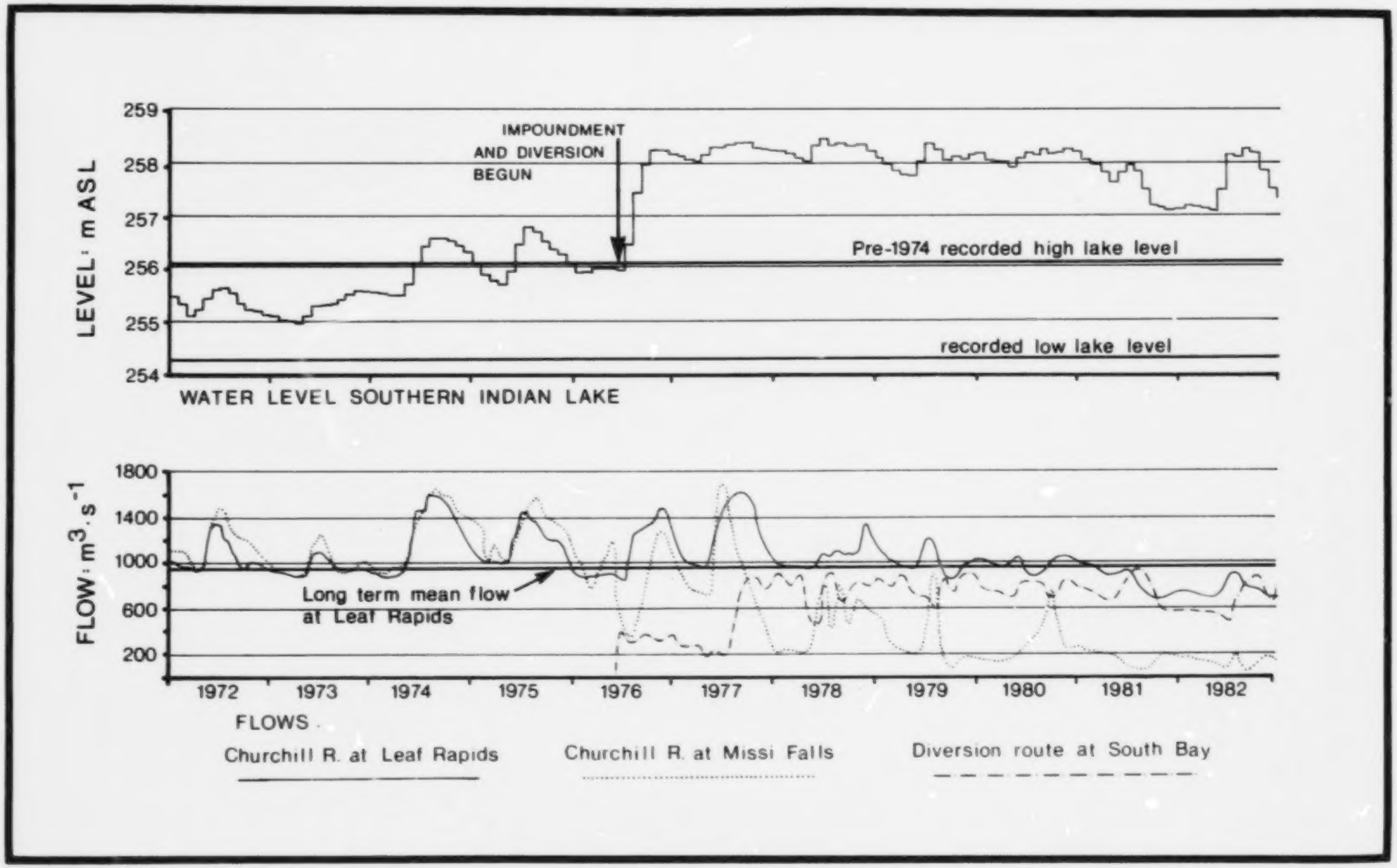
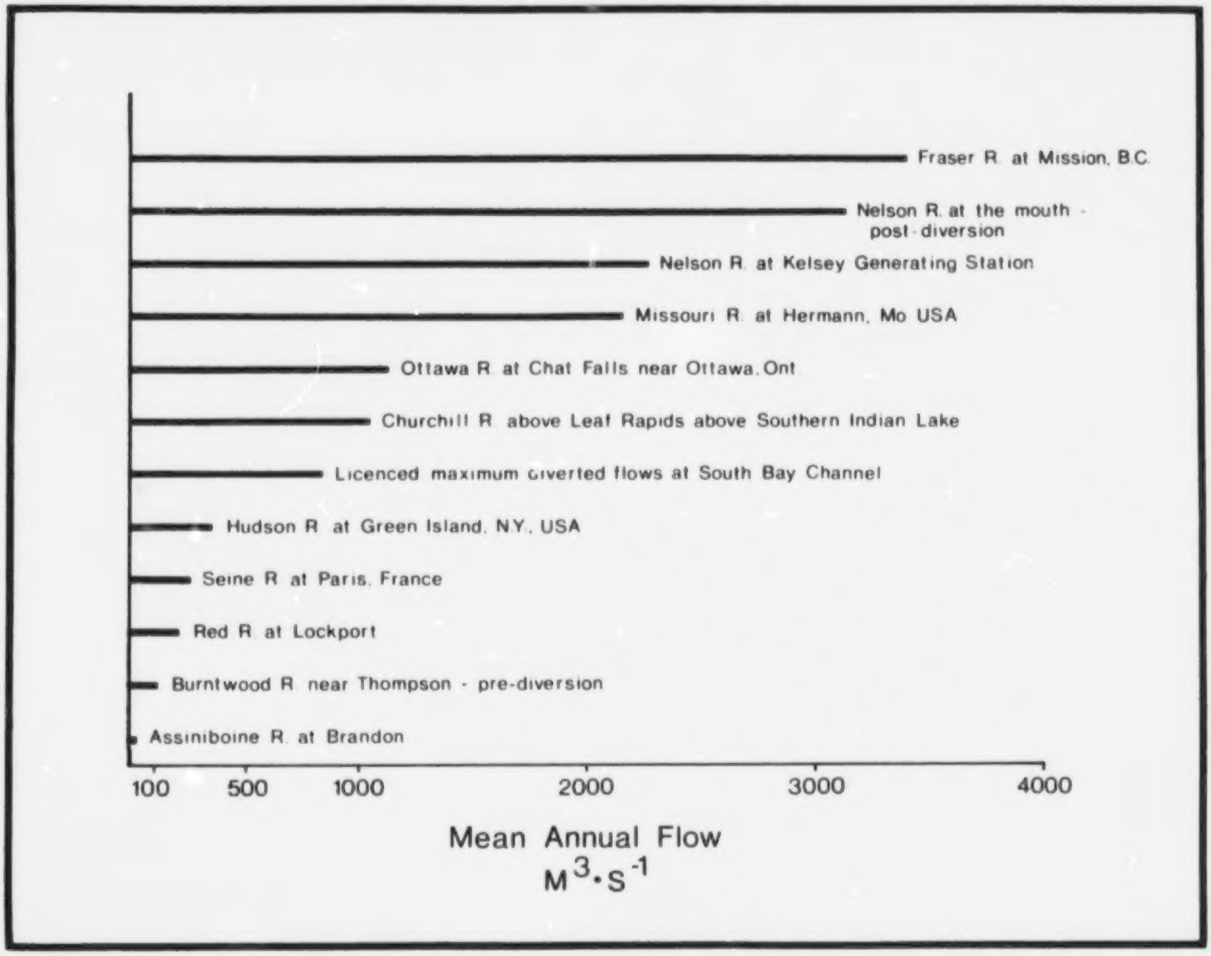


Fig. 1.4 **Diversion Flows in Comparison to Other Rivers**



THE AGREEMENT

Routine mercury sampling of the commercial fish caught from the diversion lakes in 1977/78 led to the discovery of elevated mercury levels. The mercury levels in the fish in many lakes, were too high - as high as 2 parts per million (ppm) to permit their sale in any market.

A series of discussions about the elevated mercury levels took place in 1978 and 1979 between officials of the governments of Canada and Manitoba. Two main unanswered questions emerged:

1. Was there a problem with mercury in the environment of the Churchill River Diversion; and
2. If there was a problem, what if any, was the relationship to the hydroelectric development project?

While the discussions were taking place, a claim was filed under the auspices of the **Northern Flood Agreement**. This claim, filed in April 1981, listed a number of points concerned with mercury in the hydroelectric project area. This Agreement attempts to address some of these points of claim.

The Canada-Manitoba Agreement on the Study and Monitoring of Mercury in the Churchill River Diversion, referred to in this report as the Agreement, was signed on March 10, 1983 and made retroactive to April 1, 1982.

The four year Agreement, with a total budget of $760,000 was cost shared equally by the governments of Canada and Manitoba.

The objectives of the Agreement were:

(a) to further investigate mercury levels in water, sediments, and the aquatic food chains along the Churchill-Nelson Rivers Diversion route (including if possible, comparison with levels in a similar water body in northern Manitoba which has not received inorganic mercury loading as a result of human activity, and has not been subject to diversion);

(b) to determine the source(s) of mercury input to the diversion system;

(c) to research the mechanism(s) whereby the mercury is released and enters the food chains;

(d) to assess the significance of these phenomena to future water management activities; and

(e) to advise the public of the purpose, progress, and results of studies undertaken."

The Agreement was divided into four major programs: 1. Research and Monitoring of Source Mercury and Mobilization into Aquatic Food Chains; 2. Research and Monitoring of Mercury Bioaccumulation by Aquatic Food Chains; 3. Research and Monitoring of Mercury Levels in Fish and Animals, and Human Consequences; and 4. Information and Management. These four programs were subdivided into fourteen projects. Fig. 1.6 provides a summary of the time frame, expenditures, and implementing agency for each Agreement project. The projects' objectives, methods and results are discussed in detail in the technical appendices to this report.

Fig. 1.5 **Increases in Surface Areas of Diversion Lakes**

Name	Lake Area (ha)			
	Pre-Impoundment	Post-Impoundment	Hectare Increase	% Increase
Karsakuwigamak	1,788	3,868	2,080	116
Pemichigamau	2,018	3,937	1,919	95
Mynarski	1,963	3,631	1,668	85
Rat	7,881	16,405	8,524	108
Notigi	1,742	9,095	7,585	435
Wapisu	4,478	5,921	1,443	32
Macheewin	1,268	1,288	20	2
Osik	2,256	2,607	351	16
Footprint	2,015	2,807	772	38
Threepoint	4,494	6,626	2,132	47
Wuskwatim	4,789	4,907	118	2
Birch Tree	456	627	171	38
Mystery	904	1,166	262	78
Apussigamasi	1,433	2,235	802	64
Sipiwesk		34,570		20
Split	27,939	27,960	21	<1
Stephens	8,882	29,898	21,016	237

Fig. 1.6 **Summary of Agreement Projects**

Canada - Manitoba Mercury Agreement

Project Duration, Cost and Implementing Agency

Project Name and Number	1982	1983	1984	1985	1986	Agency	Expenditure
1.1 Mercury in Water and Sediments[4]	—	—	—			EMD	
1.2 Source and Material Balance			—	—		AO	13,656.76
1.3 Mobilization and Methylation	—					DFO	38,069.01
2.1 Experimental Ecosystems	—	—				DFO	52,540.27
2.2 Bioaccumulation Process	—					DOE	29,423.62
2.3 Biogeochemisty of Sediments	—	—	—			DOE	52,576.38
2.4 Limnological Survey (and Fish Surveys)[2]		—	—	—		DFO	210,500.00
3.1 Mercury in Fauna			—	—		EMD	26,393.00
3.2 Commercial Fish Surveys[4]	—	—	—	—	—	DFO	
3.3 Fish Surveys (and Fish Eating Guide)[3]	—	—	—	—		NR DFO	148,176.19
3.4 Mercury in Hair of Native People[4]	—	—	—	—	—	H & WC	
4.1 Public Awareness		—	—	—	—	AO	12,506.19
4.2 Study Coordination and Management		—	—	—	—	AO	109,307.59
4.3 Study Report					—	AO	39,000.00 (E)
							732,149.01

Note

1. Implementing Agency

DOE	Environment Canada
EMD	Manitoba Environmental Management Division
NR	Manitoba Department of Natural Resources
DFO	Canada Department of Fisheries and Oceans
H & WC	Canada Department of Health and Welfare
AO	Agreement Office

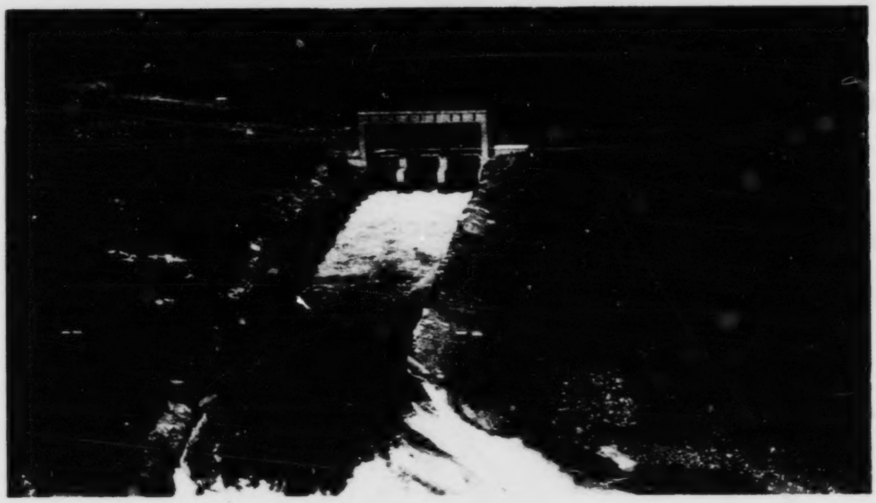
2. In 1984-85, the D.F.O. fish surveys were conducted under Project 2.4.

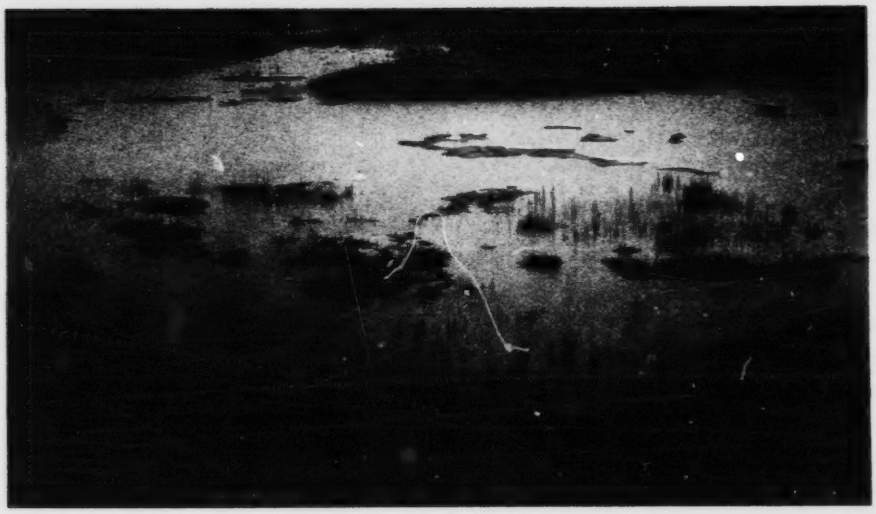
3. A decision was made not to produce a fish eating guide; instead, advice on fish consumption was provided on an individual basis, as required.

4. Projects 1.1, 3.2 and 3.4 were ongoing departmental programs which were not funded under the agreement. Project 1.1 was discontinued effective April 1, 1984.

Rat River before Flooding
Manitoba Hydro

Notigi Control Structure
Manitoba Hydro





Notigi Reservoir (1985)
Manitoba Hydro

SOURCES OF MERCURY

Mercury is widespread throughout the world occurring naturally in trace amounts in air, water, soil, plants, animals and people. Mercury can also enter the environment as a result of man's activities. Once discharged into the environment, whether from natural or man-made sources,mercury can move from one environmental component to another (Fig. 1.7).

Mercury can exist in different forms with significantly different physical and chemical properties, including differences in their toxicity to man. Interconversions between the various forms of mercury can occur. For example, it has been known since the late 1960's that inorganic mercury (the form found in bedrock) can be converted in aquatic environments to **Methyl Mercury** which is the predominant form of mercury in fish.

The determination of the source of mercury was an important consideration in understanding the reason for the elevated mercury levels in the fish along the diversion route. Particular attention was paid to potential man-made sources such as industrial, mining and agricultural activities.

There are no agricultural activities and no industrial activities with the potential for mercury discharges, such as chloro-alkali plants, located within this area. There are however, mining and smelting activities carried out within or near the Agreement area.

A number of active and inactive mines are located near Thompson, Lynn Lake, Flin Flon, and Snow Lake. Studies of mercury in water and sediment near mine sites in northern Manitoba indicate that the effect of mercury from mine tailings is limited to the small area in the immediate vicinity of the tailings. This would then suggest that mine tailings are not significant contributors to the elevated fish mercury levels observed in the Agreement area.

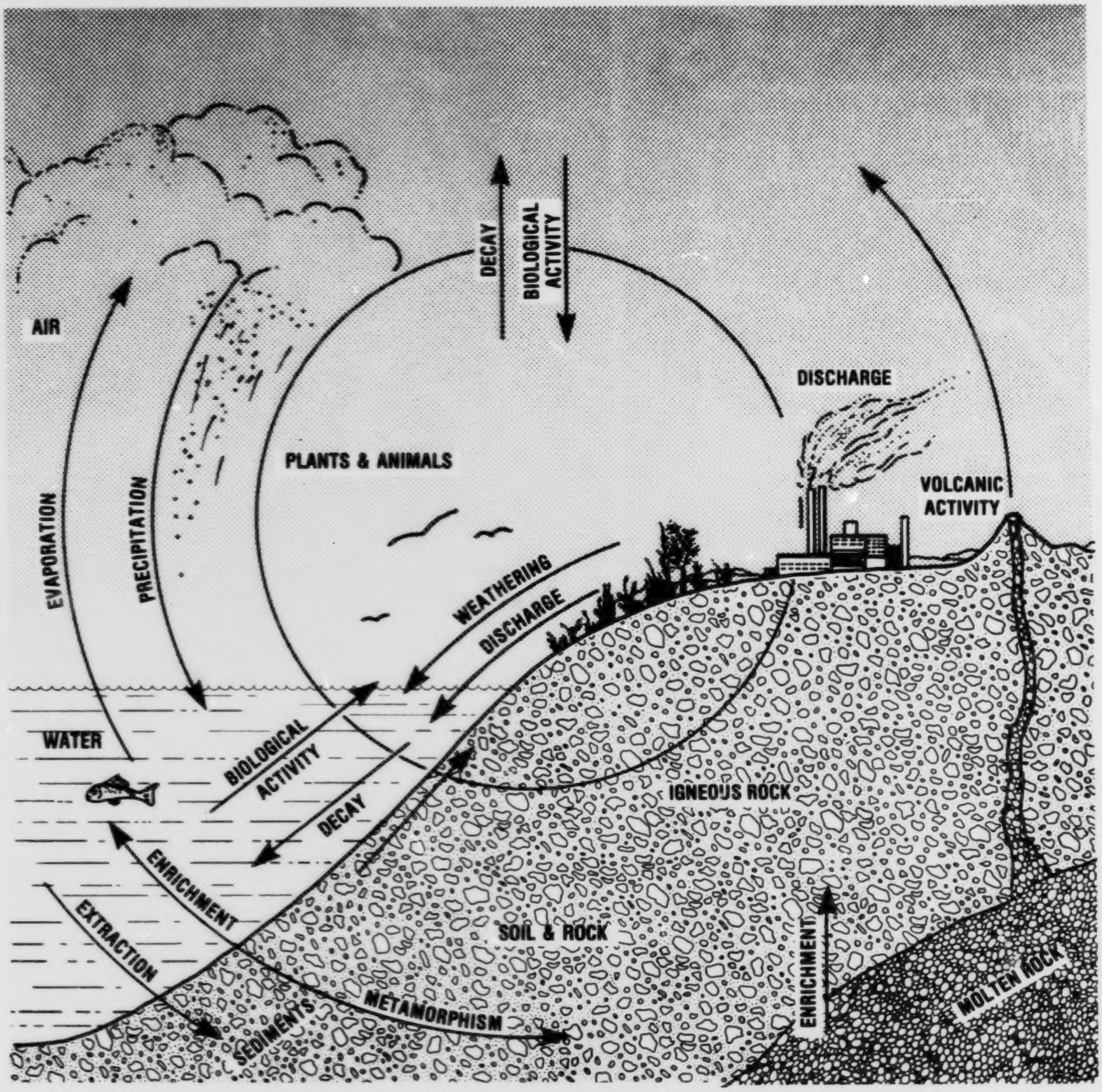
Base metal smelters, which can be important contributors of mercury to the atmosphere, are located in Thompson and in Flin Flon. Studies on the atmospheric fallout from these smelters indicate that their impact appears to be limited to an area within a few tens of kilometers of the smelters. This limited range of impact, coupled with the location of the smelters relative to the prevailing wind direction, would indicate that they had little or no impact on fish mercury levels from the diversion lakes.

Some rivers which ultimately drain into the Churchill and Nelson Rivers are known to have experienced mercury problems in the past. However there was no evidence that they have contributed to the elevated mercury levels in the fish of the Agreement area.

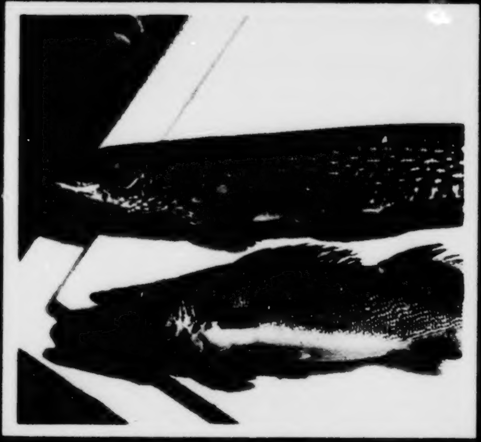
The **Volcanogenic** rocks of the region contain elevated mercury concentrations with respect to both other rock types in the study region and similar rocks in other parts of Canada. Particularly elevated concentrations occur in association with sulphide minerals.

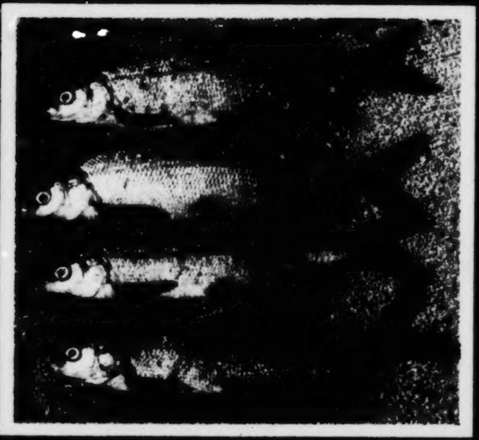
In summary, there was no evidence to indicate that significant quantities of mercury were transported into the region, either by human or natural causes. It was concluded that the elevated concentrations observed in fish were caused by factors which are internal to the region. It was further concluded that the ultimate source of the mercury was the bedrock of the region.

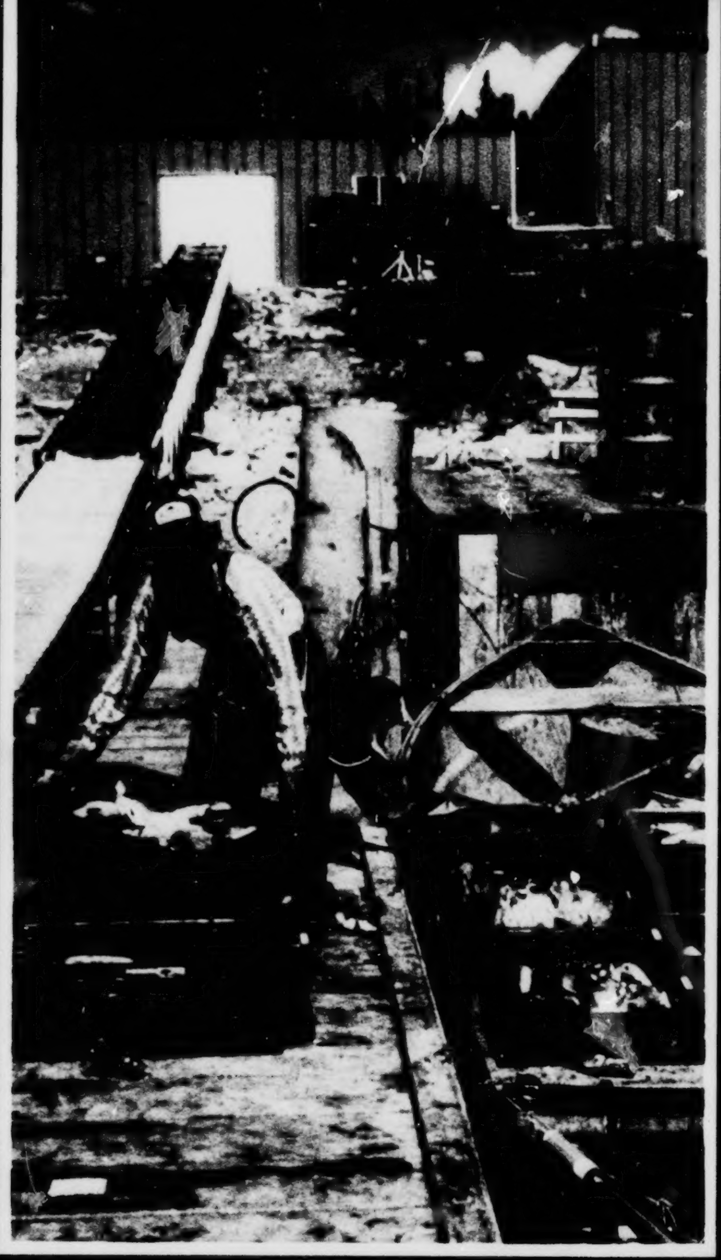
Fig. 1.7 **Mercury Cycle**


AIR
DECAY
BIOLOGICAL ACTIVITY
DISCHARGE
PLANTS & ANIMALS
VOLCANIC ACTIVITY
EVAPORATION
PRECIPITATION
WEATHERING
DISCHARGE
WATER
BIOLOGICAL ACTIVITY
DECAY
IGNEOUS ROCK
ENRICHMENT
EXTRACTION
SOIL & ROCK
ENRICHMENT
MOLTEN ROCK
SEDIMENTS
METAMORPHISM

(Mercury, which can exist in many different forms, moves between all parts of the environment - the water, air, land, and animal and plant life)







CHAPTER 2

MERCURY AND FISH

- INTRODUCTION
- MERCURY LEVELS IN FISH
- FISH CHARACTERISTICS AFFECTING MERCURY LEVELS
- MERCURY LEVELS AND FLOODING

INTRODUCTION

The fisheries resources of the lakes of the diversion route are important to the local people for two reasons; the commercial fishery provides a source of income, while the domestic fishery provides a source of good quality protein.

Two fish mercury sampling programs were active during the Agreement, the on-going commercial sampling program and the survey programs for sampling those lakes and fish species which are important to the domestic and sports fisheries. While there are some important differences between the commercial and the survey sampling programs (as noted in Appendices 14, 16 and 17), they have, individually and collectively, generated a substantial data base for analysis of mercury levels in the fish of the Agreement area.

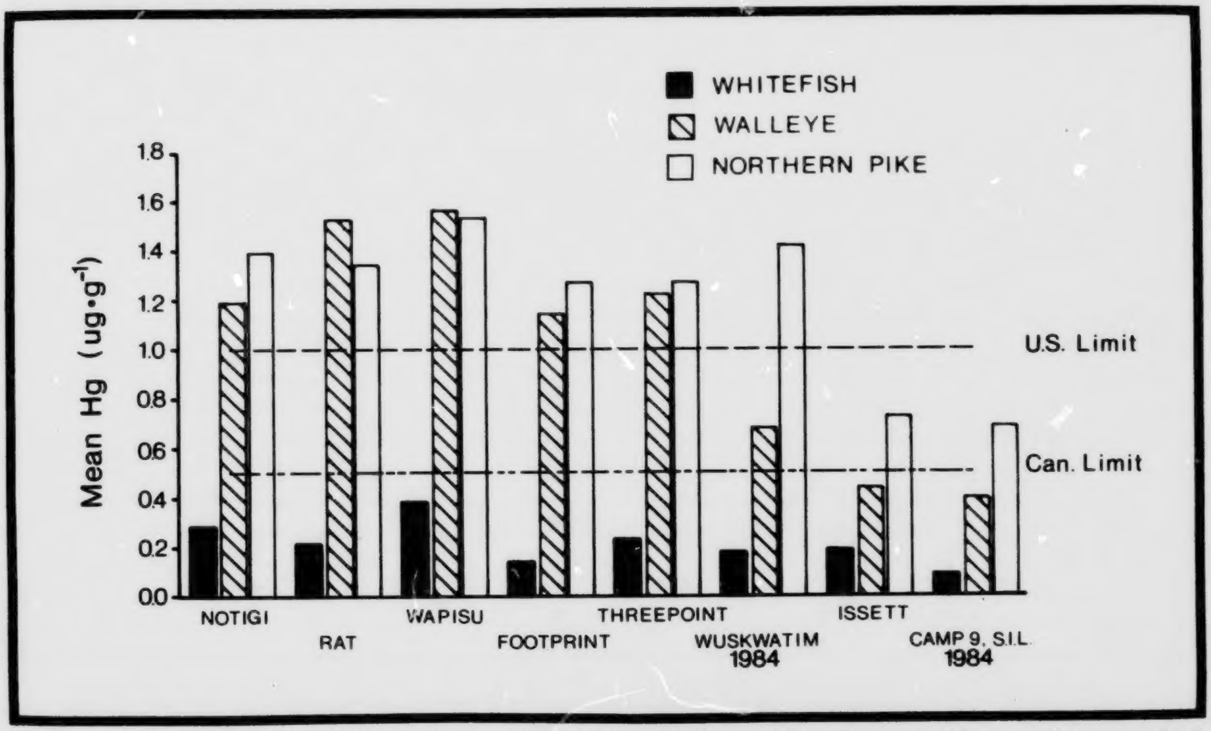
MERCURY LEVELS IN FISH

Analysis of the mercury data from the fish along the diversion route can be made relative to three parameters; 1. comparison with acceptable limits as defined by Canadian (and United States) guidelines, 2. comparison with previous levels, and 3. comparison with fish from nearby lakes unaffected by the diversion.

1. The allowable level for total mercury in fish sold in Canada is 0.5 parts per million (ppm). In the United States (U.S.), where most of the commercially harvested fish from the Agreement area are marketed, the allowable tolerance for mercury in fish was 0.5 ppm total mercury until 1979, when it was raised to 1.0 ppm total mercury. In 1984, the limit was again changed from 1.0 ppm total mercury to 1.0 ppm methyl mercury.

The most recent survey data for a number of lakes along the diversion route (Fig. 2.1) showed that the mercury levels in the whitefish were all below the Canadian limit. The mercury levels in walleye and pike from the less extensively flooded lakes were generally below the U.S. limit, while those from the more extensively flooded lakes were above the U.S. limit.

Fig. 2.1 **Mercury Levels in the Fish of Selected Diversion Lakes, 1985**



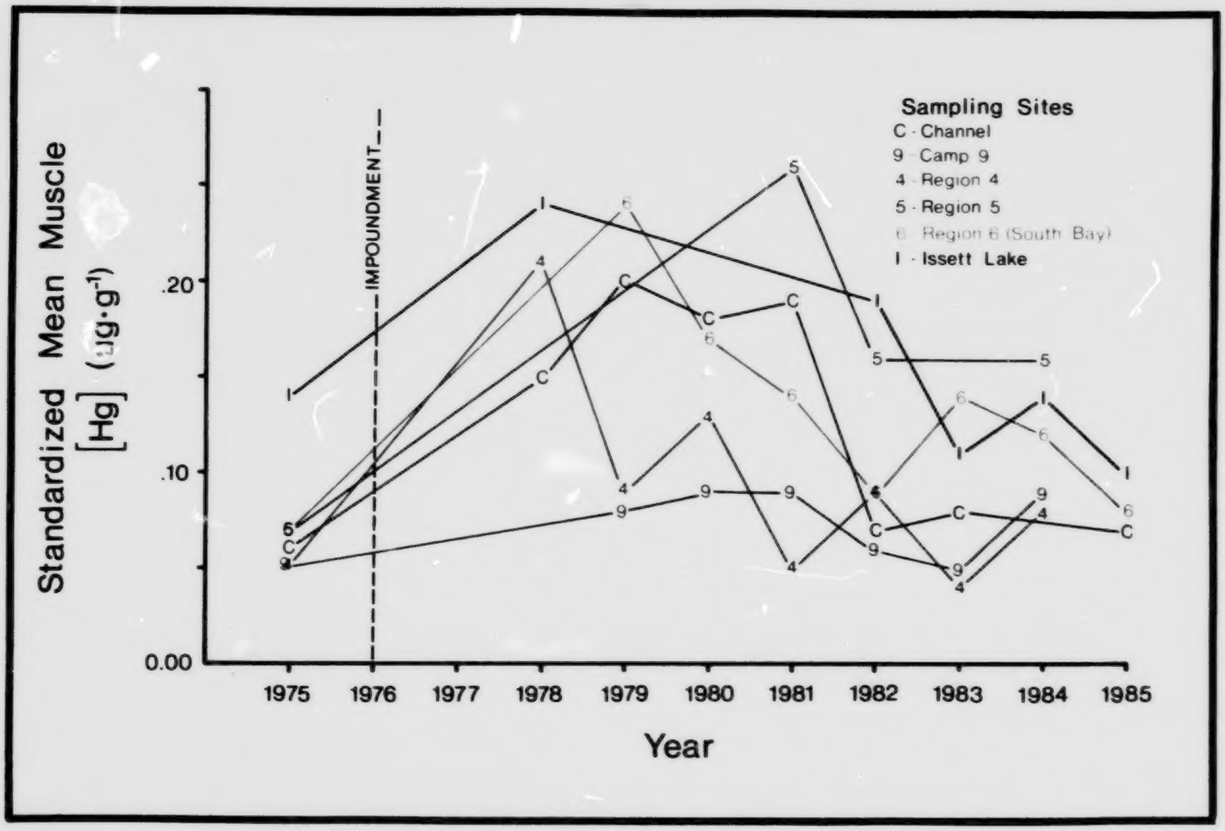
(Mercury levels in fish are reported on the basis of a standardized fork length in order to provide a consistent standard of comparison between lakes.)

2. Data from both the survey and commercial mercury sampling programs were examined to identify any trends over time in mercury levels in fish. Particular attention was given to comparisons of pre- and post-impoundment mercury levels in fish.

The only pre-impoundment survey data available are for whitefish from Southern Indian Lake and Issett Lake. These data (Fig. 2.2) show a substantial increase in the whitefish mercury levels immediately after impoundment, followed by noticeable declines for most sites over the period from 1978/79 to 1982. Mercury concentrations after 1982 were at or close to concentrations found in 1975. This suggests that mercury levels in whitefish may have returned to pre-impoundment levels by about six years after impoundment, although because only one year of sampling was conducted prior to impoundment, the pre-impoundment levels were not very precisely defined (9, 32)*.

Only post-impoundment survey data are available for walleye and pike from Southern Indian Lake and Issett Lake. These post-impoundment survey data records extend up to eight years and are plotted in Figs. 2.3, and 2.4. These data do not show any apparent trend for either the pike or walleye mercury concentrations during this time.

Fig. 2.2 **Mercury Levels in Whitefish from Southern Indian Lake and Isset Lake, 1975 - 1985**



(Mean Muscle Mercury Concentration ug.g-¹, Standardized to a Fork Length of 350 mm)

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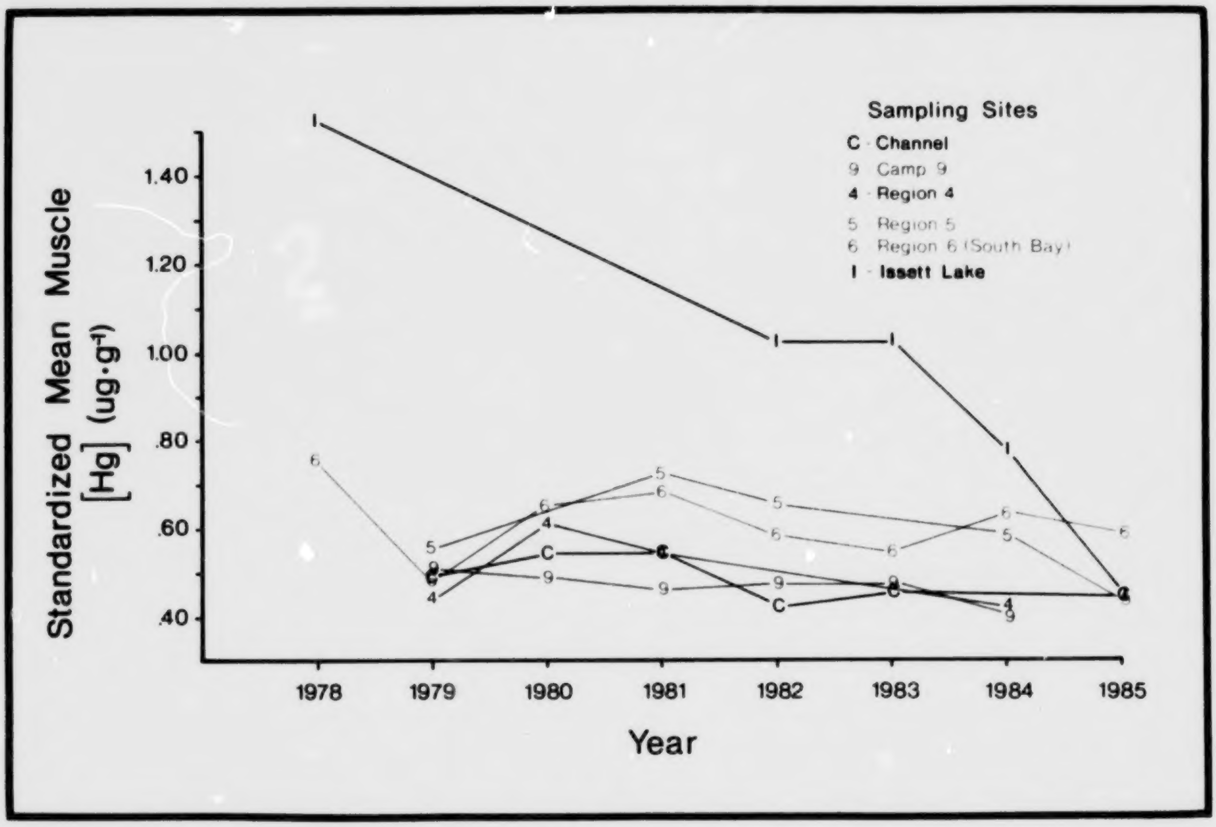
* References to material in the Agreement technical appendices are given in this format throughout the Summary Report. The first number refers to the appendix number and the second number, the page number.

Fig. 2.3 **Mercury Levels in Walleye from Southern Indian Lake and Isset Lake, 1975 - 1985**



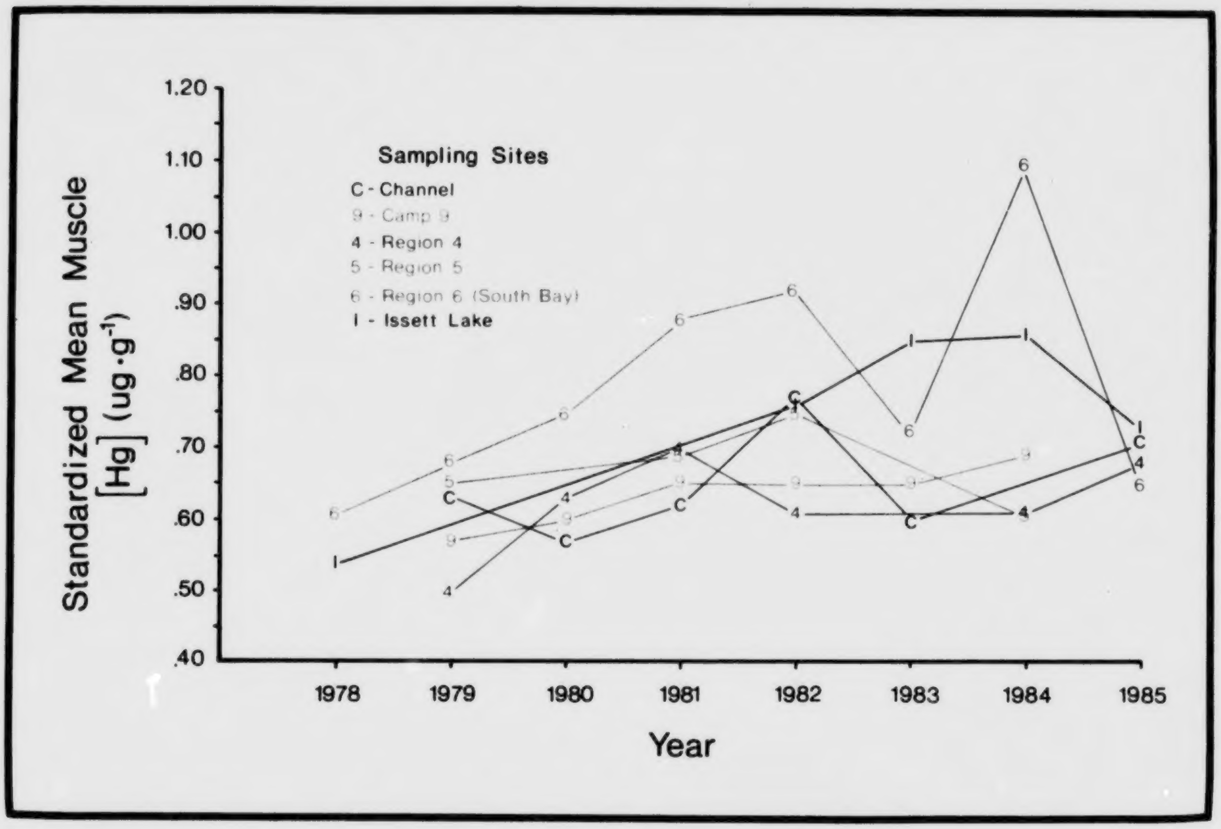
(Mean Muscle Mercury Concentration ug.g-1, Standardized to a Fork Length of 400 mm)

Southern Indian Lake supports an important commercial fishery and thus commercial mercury data for this lake are also available. Analysis of trends based on these data, however, must be viewed with caution because commercial sampling does not control fish size or specific area of the lake sampled. However, Fig. 2.5 suggests that increases in mercury levels in walleye and northern pike from Southern Indian Lake took place after impoundment. Mercury levels in pike appear to have increased steadily over the post-impoundment period, while walleye levels appear to have been stable or slightly decreasing over the same period (although commercial sample sizes for walleye were quite small in later years and conclusions based on these later data must remain tentative) (9, 35).

Survey data on mercury levels in fish from other diversion lakes were collected from 1983 to 1985. (Data from some of these lakes are shown in Fig. 2.6.) These data, along with the existing (post-impoundment) historical records were then analyzed for temporal trends. Between year comparisons of the mercury data for each lake indicated that there have been minor, if any, temporal changes in mercury concentrations or in the rates of mercury accumulation in walleye, pike and whitefish within the Agreement area over the period of record.

Similarly, fish from nearby lakes not affected, or only marginally affected, by the Churchill River Diversion have also not shown any apparent trends during the same period. The absence of any significant trend in mercury

Fig. 2.4 **Mercury Levels in Northern Pike from Southern Indian Lake and Isset Lake, 1975 - 1985**



(Mean Muscle Mercury Concentration $ug.g^{-1}$, Standardized to a Fork Length of 550 mm)

concentrations is particularly apparent in the pike from Split and Sipiwesk Lakes (Fig. 2.7) for which sample records cover a sixteen year period (17, 72).

3. Comparisons were also made between the averages and ranges in muscle mercury concentrations in fish collected from flooded lakes with those collected from non-flooded lakes within the same basin. These comparisons were conducted on nine basins in northern Manitoba. The results for the two basins along the Churchill River diversion route are shown in Fig. 2.8. While there was some overlap in ranges, the average mercury concentrations for all 3 species from the flooded lakes exceeded those of the non-flooded lakes.

These increases did not appear to be related to any higher background mercury levels; the mean mercury concentrations in the walleye and pike from the non-flooded lakes in Basins 3 and 4 were not appreciably different from the means for the non-flooded lakes in the other basins, which generally ranged from 0.3 to 0.4 ppm (17, 127). (Background mercury levels in whitefish were less than 0.1 ppm).

In summary, mercury levels in whitefish from all the lakes along the diversion route are below the Canadian limit. Mercury levels in walleye and pike from Southern Indian Lake are above the Canadian limit but below the U.S. limit.

Mercury levels in walleye and pike from the other diversion route lakes are generally above the U.S. limit. Mercury levels in whitefish from Southern Indian Lake rose immediately following impoundment; the available data

Fig. 2.5 **Mercury Levels in Commercial Fish Samples, Southern Indian Lake, 1971 - 1985**

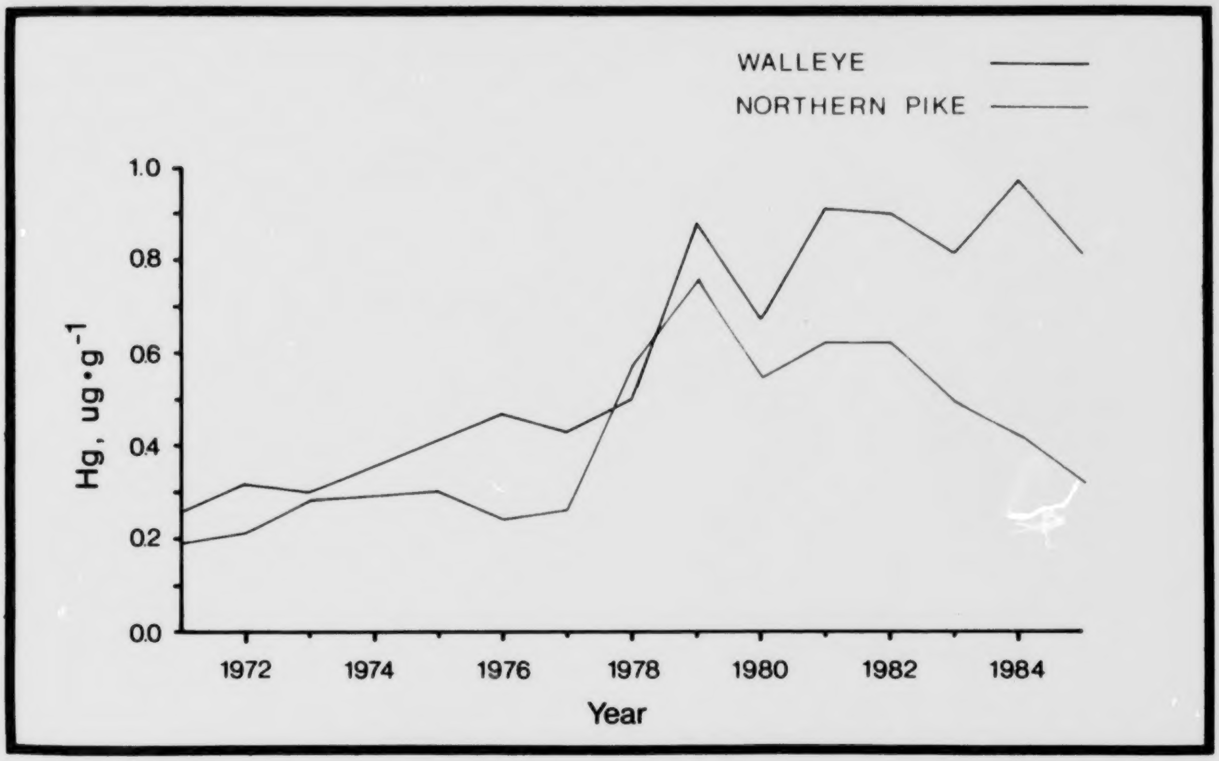


Fig. 2.6 **Mercury Levels in the Fish of Selected Diversion Lakes, 1983 - 1985**

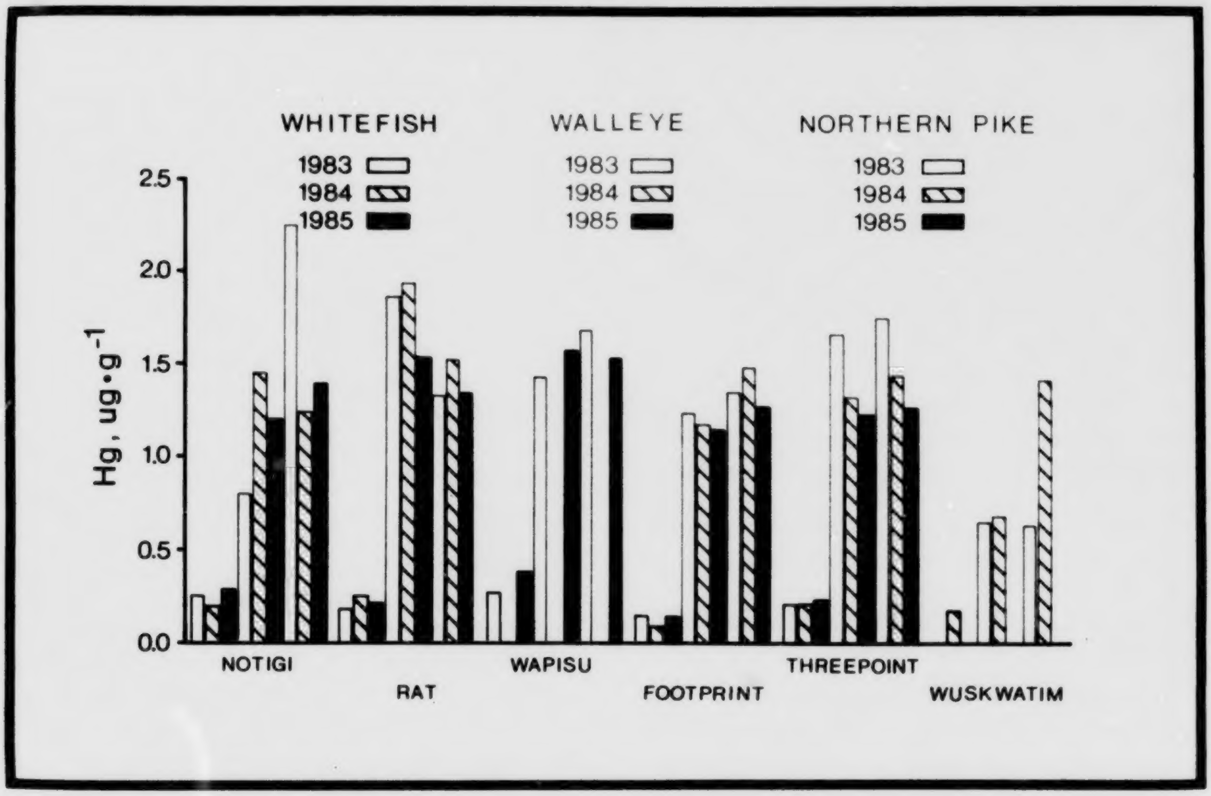
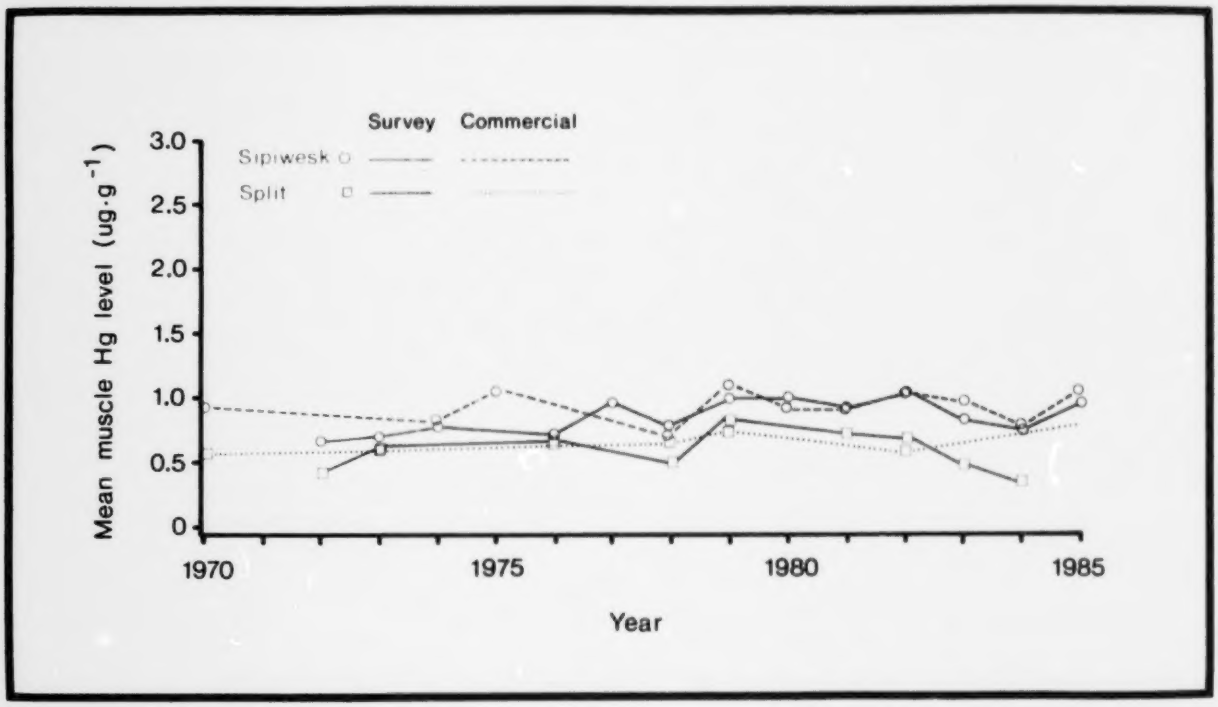


Fig. 2.7 **Mercury Levels in Northern Pike from Sipiwesk and Split Lakes, 1970 - 1985**



(Annual Mean Muscle Mercury Concentrations)

suggests that the mercury levels in pike and walleye in Southern Indian Lake also rose after impoundment. Lack of pre-impoundment data for most of the other diversion route lakes prevents comparison with post-impoundment data.

Mercury levels in whitefish from Southern Indian Lake have declined and now appear to be at pre-impoundment levels. There are no other apparent post-impoundment trends in the mercury levels in the whitefish, walleye or pike from most of the diversion route lakes. However, comparison of post-impoundment mercury levels in fish from lakes flooded by the diversion showed higher levels than in fish from non-flooded lakes within the same basin.

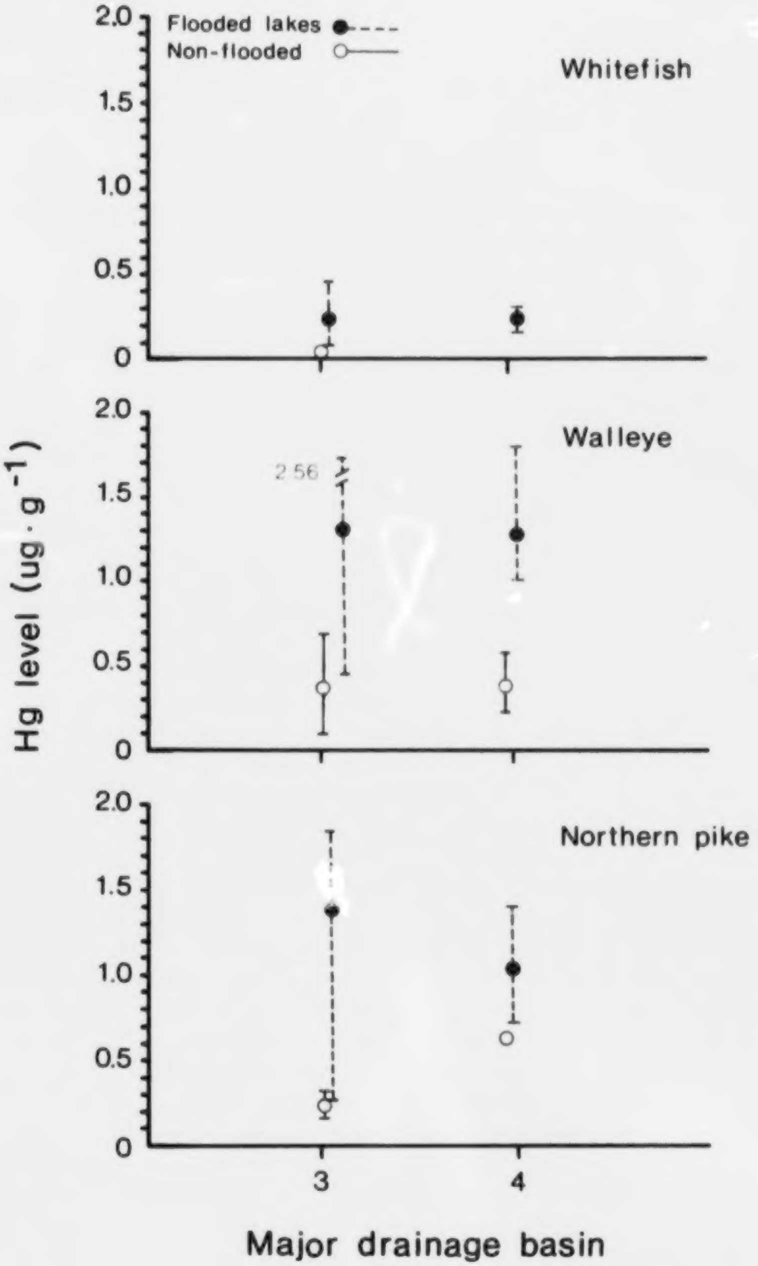
FISH CHARACTERISTICS AFFECTING MERCURY LEVELS

A number of fish characteristics were examined to determine what relationship, if any, they have to the measured mercury levels in the fish. These characteristics included length, weight and age of each individual fish as well as characteristics common to a given species.

The survey sampling programs conducted under the Agreement recorded the parameters of age, weight and length of each fish sampled for mercury. The data were then analyzed by statistical methods to determine which parameter correlated best with mercury concentrations.

Regression Analysis indicated that age generally provided the best correlation with mercury concentration for whitefish. Age also gave the best correlation for walleyes, although the correlation between length and mercury in walleyes was about as good as that between age and mercury. However, length was the best correlate with mercury in the large majority of cases for northern pike. Weight was the

Fig. 2.8 **Fish Mercury Concentrations from Flooded and Non-Flooded Lakes from the Same Drainage Basins**



(Basin 3 - Churchill River Diversion from South Bay, Southern Indian lake to Opegano Lake. Basin 4 - Churchill River Diversion from Opegano Lake to Split Lake. Mercury Concentrations have been corrected to standard sizes.)

poorest correlate with mercury for both northern pike and walleyes, but for whitefish it was a much better correlate than length (17, 92).

The relationship between fish age or size and mercury uptake appears to be primarily exponential for walleye and northern pike and linear for whitefish. The exponential character of the mercury versus size or age relationship is due to the much lower mercury concentrations in the smaller, younger fish (17, 94). The lower mercury concentrations in the young fish may be a consequence of several factors such as differences in diet and metabolic rates, as well as differences in exposure times between young and adult fish.

In summary, the older and usually larger fish (as measured by length for pike and walleye and weight for whitefish) will have higher mercury levels than the younger and smaller fish of the same species.

Regression analysis of mercury content and fish size suggested that accumulation rates for whitefish, walleye and northern pike were species dependent and largely constant from year to year and from lake to lake within the Agreement area. The concept of a fixed rate of mercury accumulation for a given species under given environmental conditions has also been suggested by the results of other recent laboratory and field studies on mercury uptake (17, 100).

The different rates of mercury accumulation were manifested in the differences in the mercury levels between species (Fig. 2.9). Elevated mercury concentrations in fish from the Agreement area were evident primarily in the large **Piscivorous** species i.e. northern pike, walleye, and sauger. (Whitefish are not considered a piscivorous species). The higher mercury content of large piscivores has often been attributed to the **Bio-Magnification** of mercury at successively higher levels of the aquatic food chain (17, 91). However, other factors such as differences in food consumption rates, food conversion efficiencies, mercury elimination rates, and fish activity levels may also be important.

Habitat preference may also be a factor. Northern pike prefer inshore areas, whitefish prefer offshore areas, and walleye move between the inshore and offshore areas. As discussed in Chapter 3, the inshore areas are the locations of the highest rates of **Microbial Methylation** of mercury. Thus differences in mercury levels between species may be, in part, the result of the differing amounts of time they spend in the areas of greatest methyl mercury availability.

In summary, mercury levels are higher in the piscivorous species of fish than in the non-piscivores. A number of factors such as diet and habitat preference may account for this difference between species.

MERCURY LEVELS AND FLOODING

No direct relationship between the extent of flooding, either in absolute terms or on a percentage basis, and the level of mercury contamination in whitefish, walleye or northern pike was apparent. However, when both the level of mercury contamination and the degree of flooding were ranked and regression analysis applied to these ranked values (Fig. 2.10), significant rank correlations between flooding and mercury contamination were obtained for all three species (17, 82).

Fig. 2.10 indicates that the level of mercury contamination in fish from a lake was directly correlated to an increase in the area flooded around the lake. However, this was not a close or well-defined relationship since there were a number of outlying points (17, 82). Included among the outlying points were some with an unusually high mercury content relative to the extent of flooding and conversely there were some with a low level of mercury contamination despite extensive flooding. Thus, while there does appear to be a relationship between the amount of flooding and the mercury levels in fish, it is obvious that flooding is not the sole determining factor. Hence, it is not possible to predict, with a reasonable degree of confidence, the level of fish mercury concentrations in impounded waters based solely on the extent of flooding (17, 112).

While it was not possible to derive a precise predictive equation, the empirical results of the fish mercury sampling program did allow for a crude estimate of the amount that fish mercury levels would be elevated given varying degrees of flooding. The relationships of the extent of

Fig. 2.9 **Mercury Concentrations in Various Fish Species, 1983 - 1985**

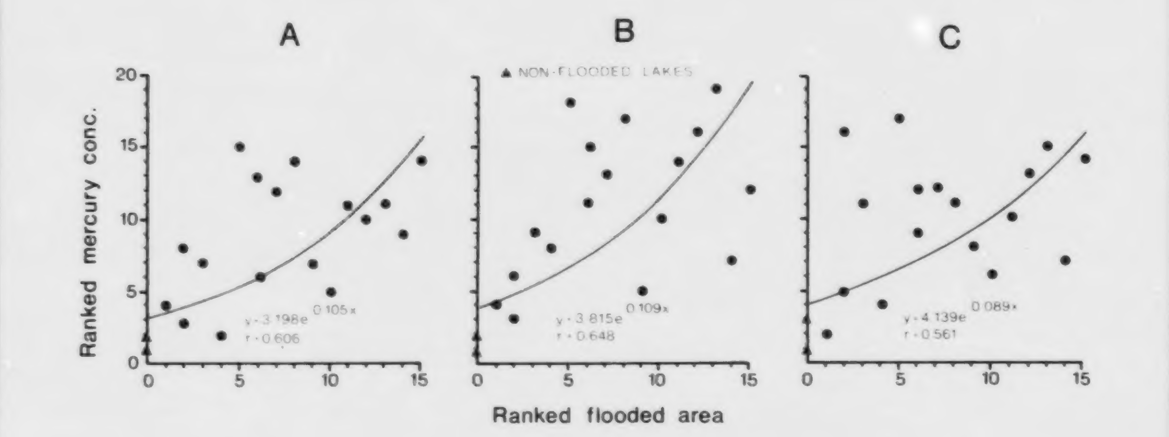
Class Interval (Mercury $\mu g.g^{-1}$)	L. whitefish	Cisco	Walleye	Sauger	Y. perch	N. pike	W. sucker	LN sucker	SH redhorse	Burbot	FW drum
0.00 - 0.09	7	5			2		2	1	1	1	
0.10 - 0.19	4	2	1		5	1	5	3	1	4	1
0.20 - 0.29	15	7	1		5	2	3	5		2	2
0.30 - 0.39	3	10	2		6	1	6	3	1	5	
0.40 - 0.49		4		3	3	2	6	2	3	4	
0.50 - 0.59			2		1	2	1	3		1	
0.60 - 0.69			2	2	1			1			
0.70 - 0.79			3	1			1				
0.80 - 0.89			3			1					
0.90 - 0.99			1								
1.00 - 1.09			3			2					
1.10 - 1.19			4	1		4					
1.20 - 1.29			1	1		3					
1.30 - 1.39			2	1							
1.40 - 1.49			2	2		3					
1.50 - 1.59			3			2					
1.60 - 1.69				5		3					
1.70 - 1.79			3	2		2					
1.80 - 1.89			1	1		1					
1.90 - 1.99						2					
2.00 - 2.09				1							
*											
2.80 - 2.89				1							
	(1.07)	(1.02)	(4.10)	(2.85)	(0.74)	(4.73)	(0.92)	(0.91)	(0.68)	(1.10)	(0.25)

(Distribution of sample means of mercury concentrations from the Churchill River Diversion and Nelson River Lakes.
Number in brackets are the maximum mercury concentrations found in individual fish.)

flooding to the length adjusted mercury concentrations in whitefish, walleye, and pike are shown in Fig. 2.11. All three species showed an initial rapid increase in mercury concentrations, followed by a tendency for mercury concentrations to level off with increasing extent of flooding. While whitefish mercury levels levelled off at approximately 30 percent flooding, walleye and pike did not do so until approximately the 100 percent flooding stage.

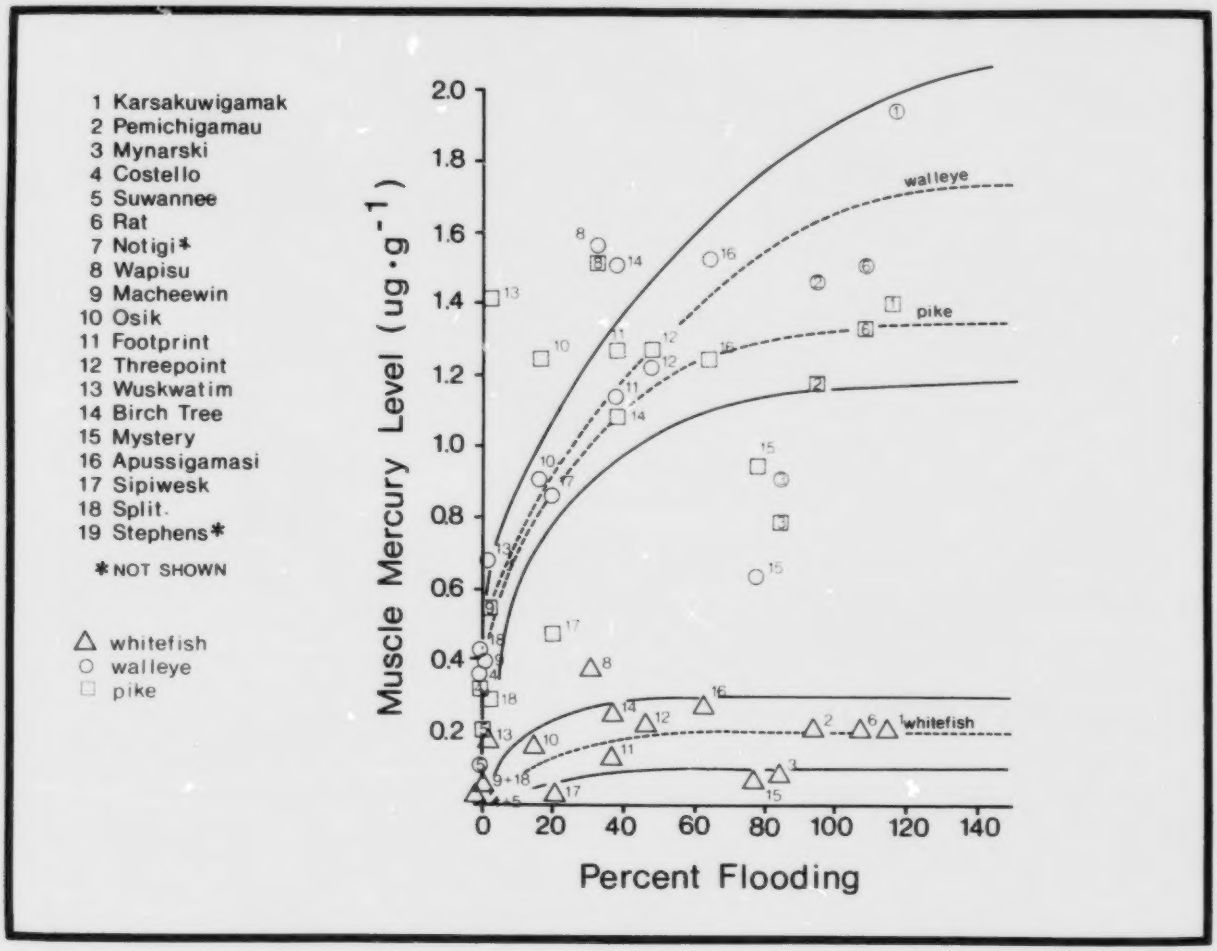
It is important to remember that Fig. 2.11 provides only a crude estimate. Its use should be limited to lakes with environmental conditions similar to those from which the data were generated. However, it may prove a useful tool in estimating the extent that fish mercury levels may increase as a result of future hydroelectric developments planned for the Churchill River diversion route, if the expected degree of flooding on each of these reservoirs is known.

Fig. 2.10 **Relationship of Ranked Flooded Areas to Ranked Mercury Concentrations in Fish**

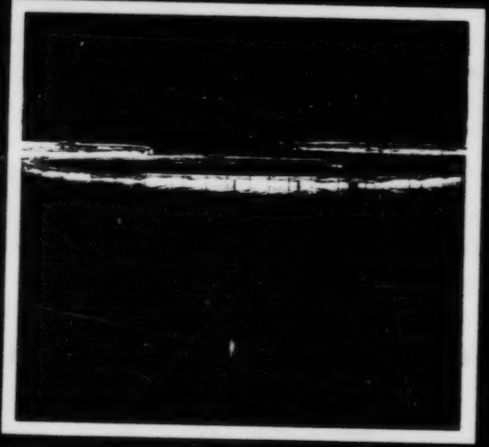


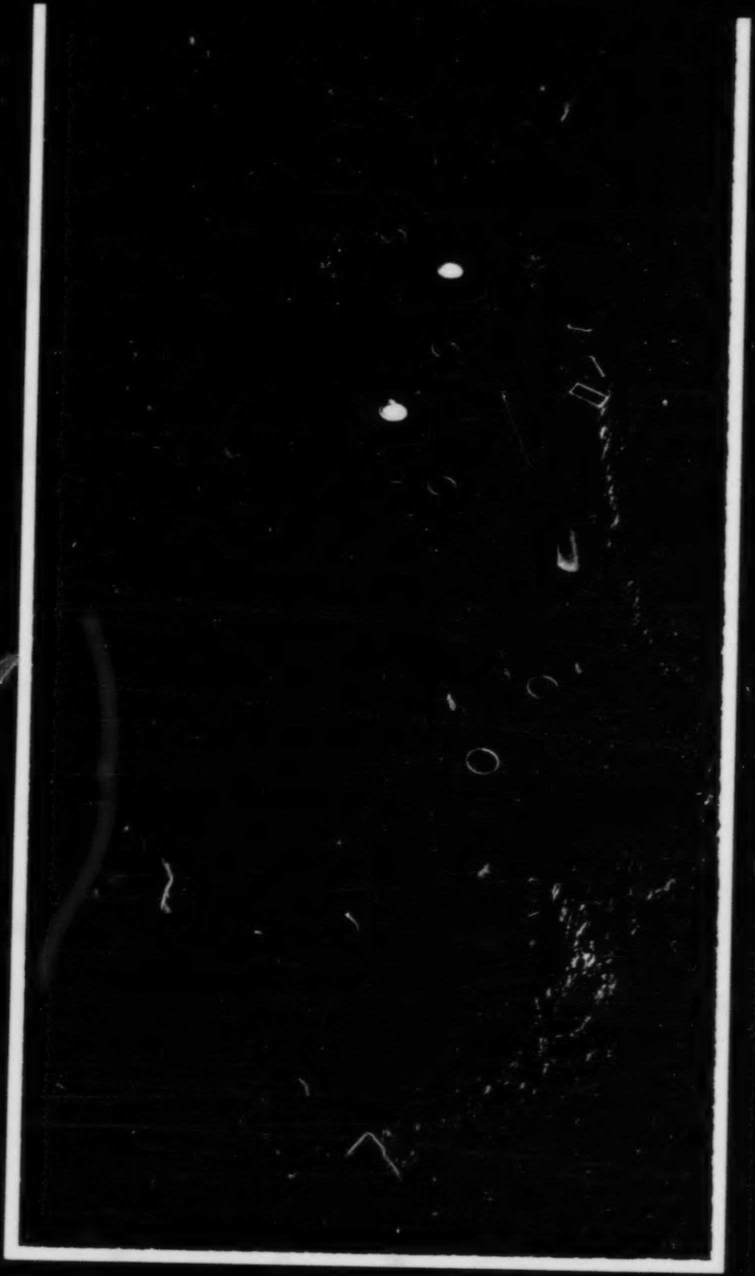
(Mercury concentrations are length corrected. Fish are from lakes in the Churchill River Diversion and Nelson River.)

Fig. 2.11 **Relationship between Fish Mercury Concentrations and the Extent of Flooding in each Lake**



(Length adjusted Muscle Mercury Concentrations for fish from lakes on the Churchill River Diversion route and Nelson River.)





CHAPTER 3

MERCURY METHYLATION AND DEMETHYLATION

- MERCURY
- METHYLATION PROCESSES
- METHYLATION RATES AND MERCURY IN FISH
- METHYLATION - DEMETHYLATION (M/D) RATIO
- MERCURY UPTAKE BY FISH

MERCURY

Mercury in Sediments, Soil, and Vegetation

As noted in Chapter 1, possible **Anthropogenic** sources of mercury to the Agreement area were investigated and subsequently determined not to have been a factor in the elevated mercury levels in fish. Measurements of the natural concentrations of mercury in the Agreement area were also investigated (Fig. 3.1) to determine if levels were elevated relative to other nonpolluted areas and whether any differences existed on either a spatial or temporal basis.

Mercury concentrations in sediments were generally found to be close to the detection limit and virtually all (12, Tb 7) were within the range of reported levels for other natural, unpolluted areas (3, 16). Soil mercury concentrations were also similar to other natural areas. Upper soil layers were found to average about $0.1 ug.g^{-1}$ mercury dry weight, while clay subsoils averaged about $0.03\ ug.g^{-1}$ mercury dry weight (3, 16). Similarly, mercury analysis of unflooded vegetation from Wupaw Bay, Southern Indian Lake, were generally less than $0.03\ ug.g^{-1}$ for trees and approximately $0.05\ ug.g^{-1}$ for mosses (4-Tb7).

There were no discernible patterns of regional variation in the mercury content of unflooded soil or in sediments (3, 9). Temporal analysis of sediment samples taken at sites within or near the Agreement area also did not show any trends (1, 16).

Although the soils and sediments have relatively low mercury concentrations, the flooded soil and vegetation have mercury concentrations averaging approximately three times higher than the preexisting lake sediment (3, 17). Experiments were undertaken to determine if there was any depletion of this mercury from flooded soils. There was little indication of any significant changes in the concentration of mercury levels in flooded soil five years after impoundment, but interpretation of the available data was difficult because of the large amount of intra- and inter-site variability (3, 13). Similarly, an experiment whereby moss was placed at lake bottom sites and subsampled over a period of 13 months did not show any appreciable change in the mercury concentration of the moss (Fig. 3.2).

It should also be noted that Agreement studies confirmed the lack of a relationship between mercury levels in sediments and mercury levels in fish that has been reported in other studies (3, 18). This finding points to the overriding importance of modifying factors in the determination of fish mercury concentrations.

In summary, the mercury levels in the soil, sediment, and vegetation of the Agreement area are normal as compared with levels in other non-polluted regions. There was no evidence of temporal or spatial trends in mercury levels, of any depletion in mercury levels following flooding, or of any relationship between mercury levels in fish and in sediments.

Mercury in Water

Measurements of mercury concentrations in water yielded results which were generally below the detection limit. Mercury concentrations in **Plankton** were therefore used to provide a relative measure of the mercury concentrations in water. It was found that the mercury levels in the control lakes and flooded reservoirs were similar and were comparable to those levels reported for other uncontaminated systems. Thus, there is no evidence that flooding has caused an increase in mercury concentrations in water in any of the reservoirs studied nor were the concentrations unusually high (12, 17).

One interesting observation was a distinct seasonal fluctuation of plankton mercury concentrations found in all of the lakes and reservoirs sampled in 1984 (Fig. 3.3). It was postulated that the observed influx of inorganic mercury to the water in August resulted from circulation within the sediments and a flushing of dissolved constituents caused by fall cooling (12, 19). The 1985 survey however did not show this distinct seasonal fluctuation, possibly because there was no distinct cooling period at any of the sites during the period of observation (13, 13).

Fig. 3.1 **Mercury Levels in Environmental Components**

Component	Selected Locations at Selected Years Within Agreement Area				Agreement Range	Other Unpolluted Areas	
Sediment							
(µg/g dry weight)	Southern Indian Lake[1]	<0.03-0.06	Footprint Lake[1]	<0.03-0.09	<0.02[1]-0.09	Yukon Lakes	<0.01-0.05
	(near South Indian Lake)		(near Nelson House)			Northern Quebec avg.	0.068
	Granville Lake (mean)	0.024	Notigi West (mean)	0.028	0.009-0.060	Pristine Wisconsin Lakes	0.05-0.15
	East Mynarski (mean)	0.020	Wupaw Bay (mean)	0.053			
Soil	unflooded upper soil						
(µg/g dry weight)	Footprint Lake (mean)	0.079	Notigi West (mean)	0.052	0.052-0.174	Avg. earth's crust	0.08
	Granville Lake (mean)	0.122	Wupaw Bay (mean)	0.102		Finland upper soil	0.10
	unflooded clay subsoil					Illinois unflooded soil	0.04-0.15
	Footprint Lake (mean)	0.037	Notigi West (mean)	0.038	0.022-0.056	Finland deeper soil	0.02
	Granville Lake (mean)	0.027	Wupaw Bay (mean)	0.056			
	flooded clay subsoil						
	Footprint Lake (mean)	0.032	Wupaw Bay (mean)	0.041	0.016-0.043		
Vegetation	trees						
Wupaw Bay, S.I.L.	wood	<0.005-0.015	twigs	<0.005-0.070	<0.005-0.070		
(µg/g)	bark	<0.005-0.029	foliage	<0.005-0.065			
	moss	0.022-0.13			0.022-0.13		
Water							
(µg/g)	Southern Indian Lake[1]	<0.02	Footprint Lake[1]	<0.02	<0.02[1]		
Net Plankton							
(µg/g dry weight)	Methyl Bay[2]	0.026-0.200					
	Granville Lake[2]	0.042-0.084			0.026[2]-0.200	Cedar Lake[2], Manitoba	0.042-0.210

Footnotes

[1]. Source Project 1.1

[2]. Source Project 2.4

All other data are from Project 1.3

Detection Limits

Project 1.1	20 ng/g sediment	20 ng/l water
1.3 and 2.4	5 ng/g sediment	5 ng/l water

Fig. 3.2 **Mercury Concentrations of Moss Samples**

Site	Date	Total Mercury ($ng.g^{-1}$)	Methyl Mercury ($ng.g^{-1}$)
Initial Samples	July 15, 1982	80	7
		90	3
		70	4
		60	6
Lake	August 10, 1982	80	9
		80	<1
	October 18, 1982	90	<1
		110	<1
		70	<1
	August 26, 1983	60	3
		70	6
Bog	August 10, 1982	120	3
		90	8
	August 26, 1982	80	3
		80	1

Bioavailable Mercury

All of the data shown in Fig. 3.1 were measurements of total mercury concentrations. As was noted in Chapter 1, total mercury is composed of a number of different forms of mercury. The methyl mercury form is the one of interest in this report since it is the predominant form of mercury found in fish.

The overwhelming souce of methyl mercury is the methylation of the mercuric ion (Hg^{2+}) by **Microbes** (6, 6, and 8, 24). It is not presently possible to develop a theoretical model to predict the overall effect of reservoir formation on mercuric ion or methyl mercury availability (6, 8). Thus, two empirical approaches were used in the Agreement studies to estimate the amount of bioavailable or methylatable mercury.

One approach was based on defining the amount of mercury potentially available for methylation as those forms of mercury which are weakly bonded to the surfaces of sediment particles. Three different parameters were used in a series of laboratory experiments to estimate bioavailable mercury using this premise. The parameters were subsequently used as a part of a compound variable, which when plotted against the mercury content of walleye, gave significant correlations (8, 46). It should be noted however that significant correlations were obtained only for walleye. In addition, the individual variables when tested separately gave decidedly insignificant correlations, demonstrating that it was the combination of factors, not any one of them alone that controlled mercury uptake by walleye (8, 47).

A more direct estimate of methylatable mercury was obtained by adding a **Radioactive** spike of mercuric ion ($^{203}Hg^{2}+$) to a series of **Limnocorrals.** After the addition of the spike, samples of fish (non-piscivorous) were taken from the limnocorrals at regular intervals to monitor the uptake of radioactive and total mercury by the fish. Once equilibrium was reached, it was possible to calculate the size of the methylatable pool of mercury in each limnocorral by multiplying the total mercury added in the spike by the ratio of the **Specific Activity** of the spike to the specific activity of the fish (6, Tb 1). These calculations showed that the methylatable mercury pool size was not relatable to the total mercury concentrations in fish muscle at the end of the experiments. It was concluded that the availability of mercury for methylation does not appear to be a sole controlling factor in the mobilization and accumulation of mercury in fish muscle (6, 24).

METHYLATION PROCESSES

Given that there was no evidence of any increase in the total mercury levels in the environment of the Agreement area, or of any major differences in its spatial distribution, then the elevated methyl mercury levels found in the

Fig. 3.3 A **Seasonal Mercury Levels in Net Plankton**

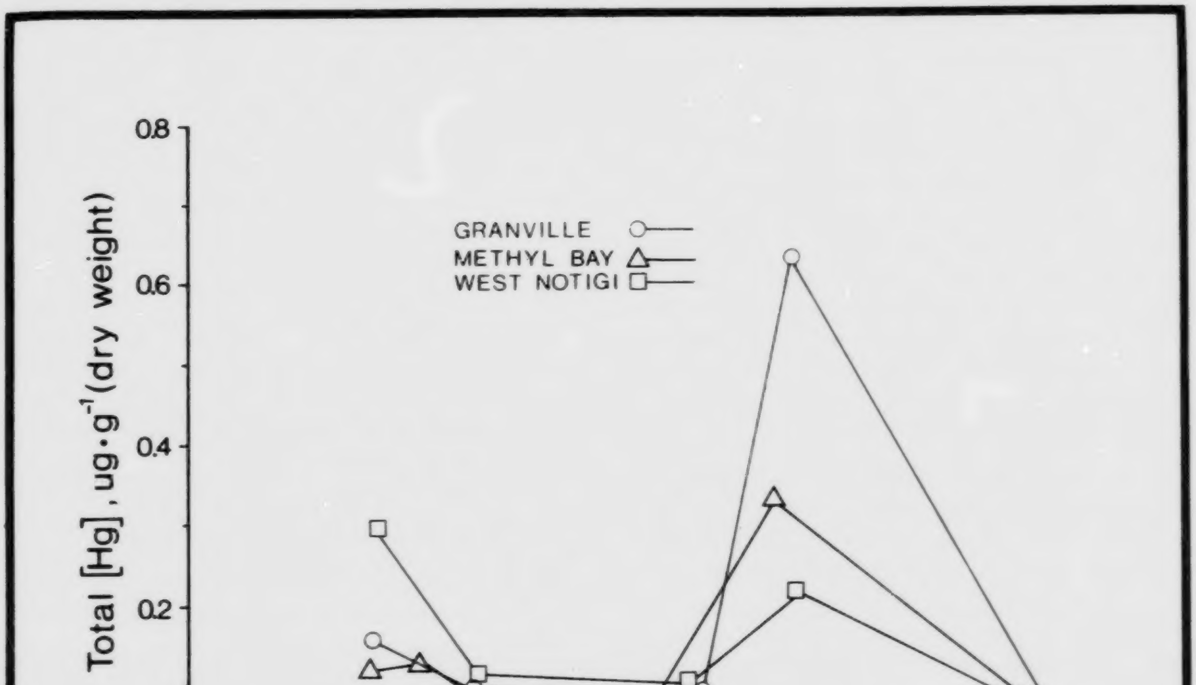
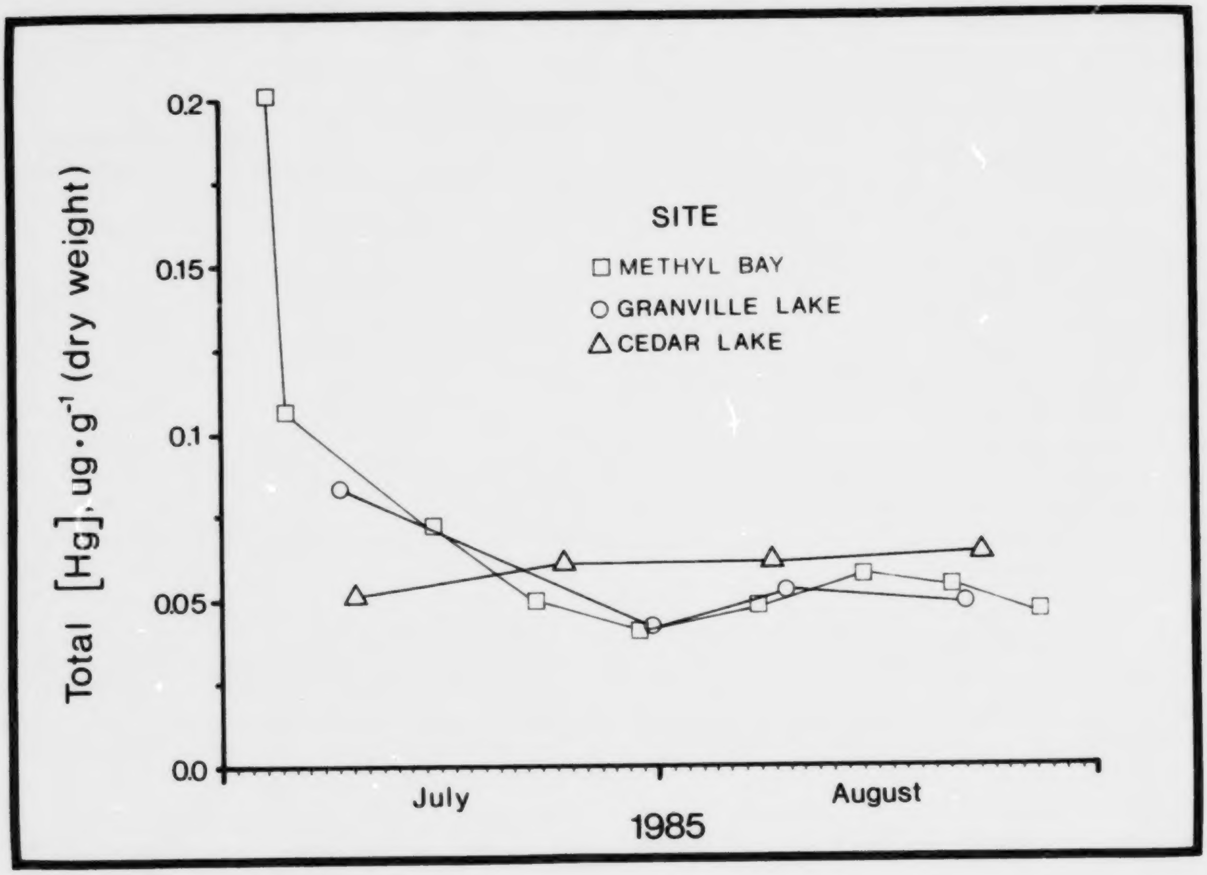


Fig. 3.3 B **Seasonal Mercury Levels in Net Plankton**



fish from the lakes along the diversion route must have resulted from processes which increased the rate of conversion of other forms of mercury to methyl mercury.

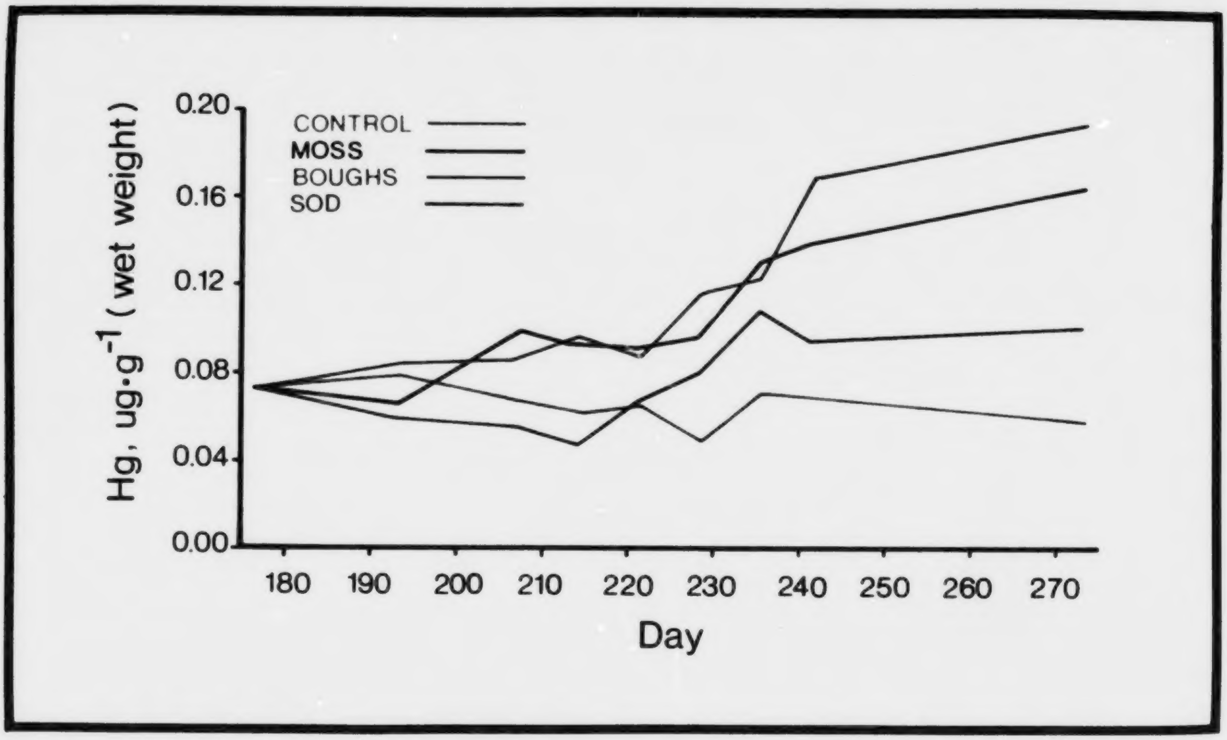
It has been known since the late 1960's that methyl mercury is produced naturally in the aquatic environment, although in small amounts. It is also known that methyl mercury is produced and destroyed by a variety of microbes. It was therefore postulated that the various physical, chemical, and biological changes that occurred following the diversion may have affected the net production of methyl mercury, primarily by affecting the balance of methylating and **Demethylating** activity in the microbial community.

A number of experiments were conducted, both in the laboratory and in the field, to test the effects of various materials and conditions on the production of methyl mercury in sediments. The materials and conditions chosen for study were selected on the basis of their relevance to known post-diversion conditions. In some cases, the effects of the various materials and conditions were determined by measuring the changes in the methyl mercury levels of experimental fish, while in other cases measurements were made directly of the amount of methyl mercury that was produced or destroyed.

Organic Materials

A series of limnocorral experiments were conducted involving the additions of moss, peat and black spruce boughs which are the most common **Organic** materials found in the

Fig. 3.4 **The Addition of Organic Materials to Limnocorral Experiments, (Natural Bottom), 1984**



Agreement area. The organic treatments were generally scaled to the flooded area to volume ratio of Notigi Reservoir (6, 12).

It was found that these materials, stimulated the **Bioaccumulation** of mercury, with black spruce boughs increasing mercury levels in fish to a greater extent than did the moss-peat material (6, 18) (Fig. 3.4). It was also found, at least in the case of moss-peat material, that the stimulatory effect is additive with a doubling of the amount or organic material added resulting in a doubling of the amount of mercury which accumulated in the fish (6, 18) (Fig. 3.5). These results were independent of the fish species used (yellow perch and emerald shiners), and of fish growth rates.

Limnocorral experiments were also conducted in which prairie sod was the organic material added. The results showed that the addition of prairie sod stimulated methyl mercury production to the same extent as did the other organic materials (Fig. 3.4). This indicates that the stimulatory effect of flooding on fish mercury levels is a general response and not limited to areas of boreal forests (6, 19). The variety of stimulatory organic materials was further confirmed by laboratory experiments utilizing freshwater algae (Fig. 3.6) which gave comparable results to those obtained from the addition of moss (8, 26).

Laboratory experiments showed that the addition of pulverized moss to sediment-water slurries promoted the net methylation of **Inorganic** mercury under both **Aerobic** and **Anaerobic** conditions (8, 25) (Fig. 3.7). Short term laboratory experiments in which the quantity of moss was varied, while keeping the incubation time constant, showed that an unlimited increase in the amount of moss did not sustain an unlimited increase in net methyl mercury production (Fig. 3.8). Other short term laboratory experiments comparing the production of methyl mercury by sediments from various Agreement area locations showed that the addition of moss actually depressed methyl mercury production for those sediments with the highest organic carbon concentrations (8, 28).

The short term of these laboratory experiments on moss in comparison to the limnocorral experiments on moss may explain the differences in their findings. Other short term laboratory experiments conducted under the Agreement indicated that moss can inhibit methylation, possibly by sequestering dissolved mercury if it is at high concentrations. This inhibition may be of short duration because as the moss is decomposed and used as a nutrient source by the microbiota, bound mercury can be released and may be methylated. (5, 21).

In summary, the limnocorral experiments and the laboratory experiments found the same basic result; namely, that a variety of organic materials stimulate the production of methyl mercury.

Inorganic Nutrients

Phosphorus and nitrogen additions were made to a limnocorral to boost **Chlorophyll** concentrations and primary productivities to levels comparable to Notigi Reservoir during and since impoundment. The rate of primary productivity in the treated limnocorrals was double the rate of productivity in control limnocorrals, yet there was no effect on the uptake of ^{203}Hg, total mercury, or fish growth (6, 20) (Fig. 3.9). It should be noted, however, that the total organic carbon loading after inorganic **Nutrient** addition was only about 10% of the organic carbon loadings of the moss-peat and spruce treatments. Therefore, the potential effect on bacterial growth and mercury methylation in the moss-peat and spruce treatments were much greater because of the greater availability of **Substrate** (6, 20).

Fig. 3.5 **The Addition of Varying Amounts of Moss to Limnocorral Experiments, (Natural Bottom), 1982**

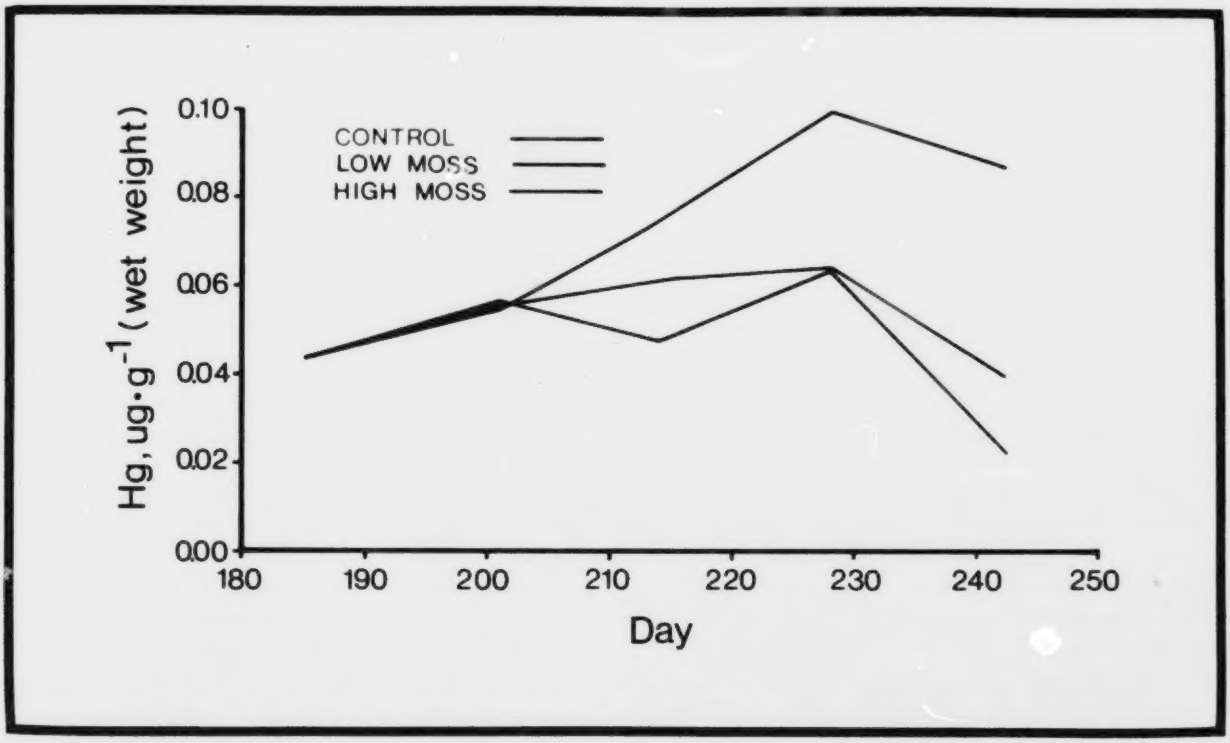
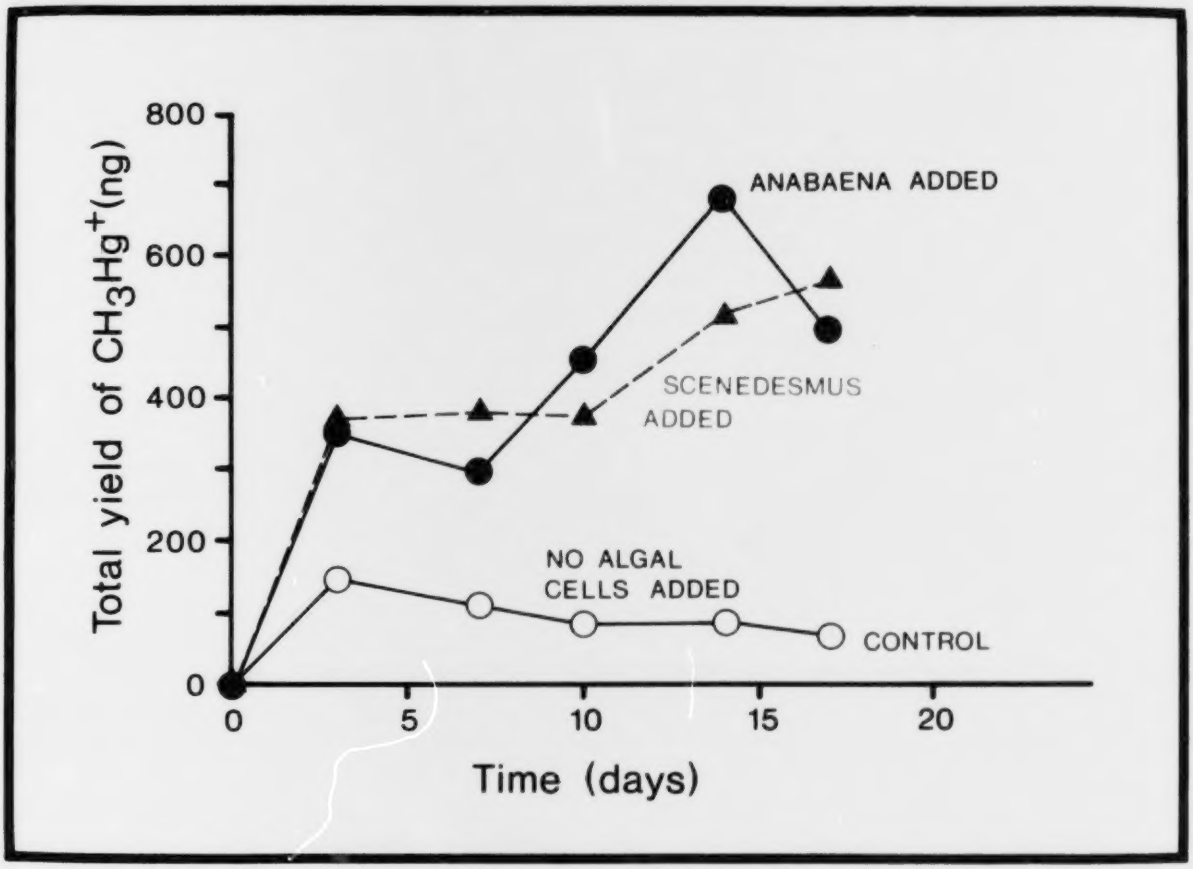


Fig. 3.6 **The Effect of Different Algae Cells on the Production of Methyl Mercury**



(Production of Methyl Mercury from Mercuric Chloride by East Mynarski sediment under Nitrogen.)

Sediment (Clay)

As noted in Chapter 1, a major difference in the water quality of Southern Indian Lake and most of Notigi Reservoir was the high suspended sediment levels in Southern Indian Lake. Eroded suspended sediments can be a source of mercury to reservoirs. An extensive series of limnocorral experiments were conducted involving the addition of suspended sediments. All the experiments indicated that suspended sediments had either a negligible or a stimulatory effect on mercury accumulation in fish (Fig. 3.10).

The addition of suspended sediments was suggested as a possible mitigation measure in the studies on the mercury contaminated English-Wabigoon system. As sediments from eroded bank clay had already been shown not to depress mercury accumulation in fish, sediments from lake bottom were added to a moss-peat treated limnocorral to test its mitigating value (Fig. 3.11). There was no effect of the suspended sediments on total mercury uptake, thus giving no evidence that suspended sediment addition has any mitigating value in one year (6, 22).

A number of laboratory experiments were also conducted to determine the effect of shoreline clay on mercury methylation. It was found that the effect of clay on methylation was complex as might be expected from the complex composition of clay. One effect, pH buffering by associated carbonate minerals, may explain why shoreline clay in its native state caused increased production of methyl mercury, for when the experimental slurries were buffered to prevent ph variations, the clay inhibited methyl mercury production slightly (8, 33).

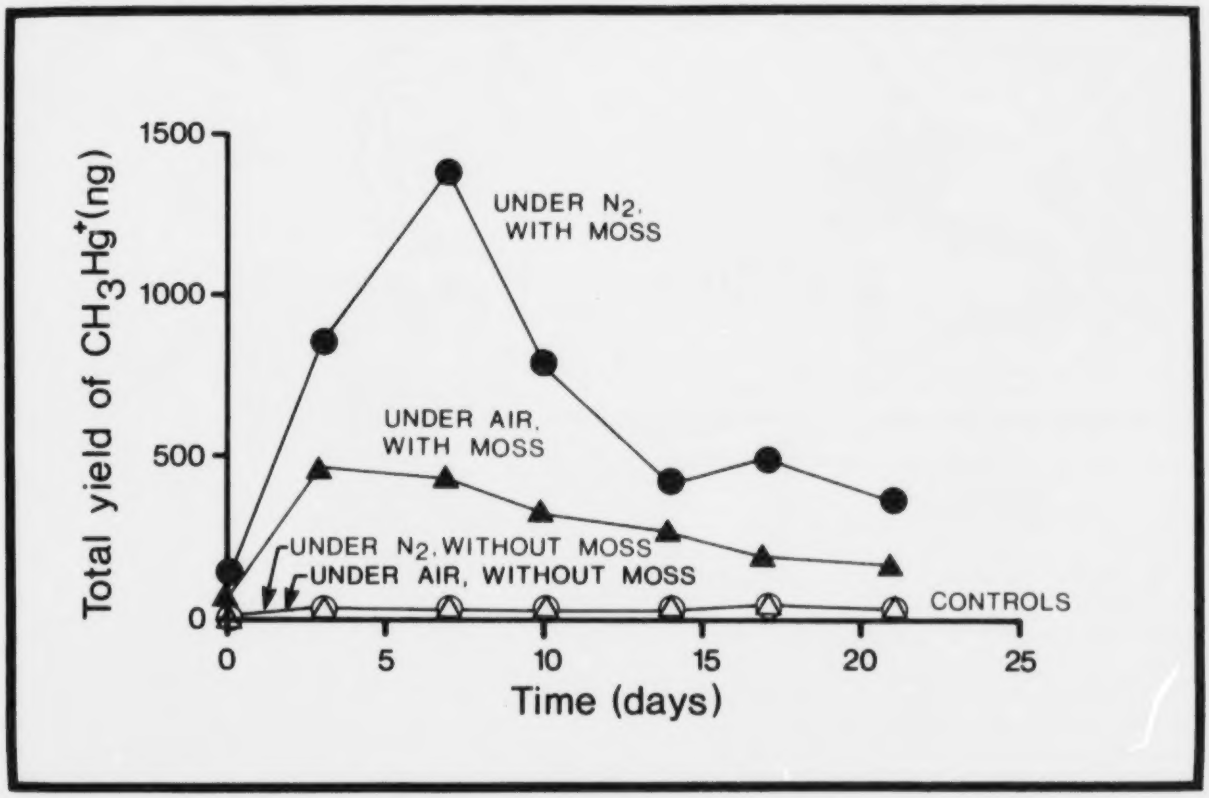
Dissolved Mercury

Laboratory experiments were conducted to test the effect of dissolved **Mercuric Chloride** on the yield of methyl mercury. Sediments were incubated, under anaerobic conditions, with and without added moss, for a fixed period of time. Different concentrations of dissolved mercuric chloride were added ranging from zero to concentrations in the slurry that were up to five orders of magnitude greater than that found in the sediment samples.

The yield of methyl mercury increased over much of the range of increasing mercuric chloride concentration in both the moss and no-moss added experiments. However, at the higher concentrations of mercuric chloride, the yield of methyl mercury in the moss added experiment declined. It was postulated that the dominant mercury methylating microbes in the nutrient enriched slurries were more sensitive to the toxic effects of mercuric chloride than were the methylators in the slurries which had not been enriched. (8, 31).

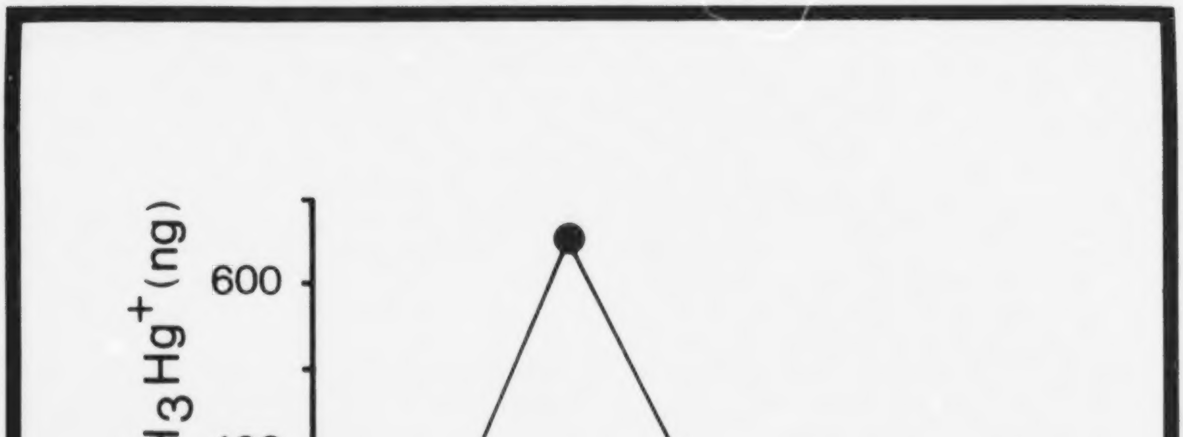
Limnocorral experiments were also run to determine whether the added mercury of the radioactive spikes had influenced the results of the previous limnocorral experiments. Mercuric chloride additions were made at three different concentrations which increased the

Fig. 3.7 **The Effect of the Addition of Moss under Aerobic and Anaerobic Conditions on Methyl Mercury Production**



(Production of Methyl Mercury from Mercuric Chloride by Calcium Carbonate buffered East Mynarski sediment.)

Fig. 3.8 **Short Term Effect of the Addition of Varying Amounts of Moss on the Yield of Methyl Mercury**



(East Mynarski Lake sediment incubated for 7.6 days under Nitrogen in the presence of Mercuric Chloride.)

total mercury concentrations in the limnocorrals by 0, 2, and 10 ng.1^{-1}. (The highest concentration of mercury created by the radioactive spike in any of the previous limnocorrals was 8ng.1^{-1}, (6, 15). A corroborative experiment was conducted by adding moss-peat material to two limnocorrals, only one of which received a radioisotope spike of mercury.

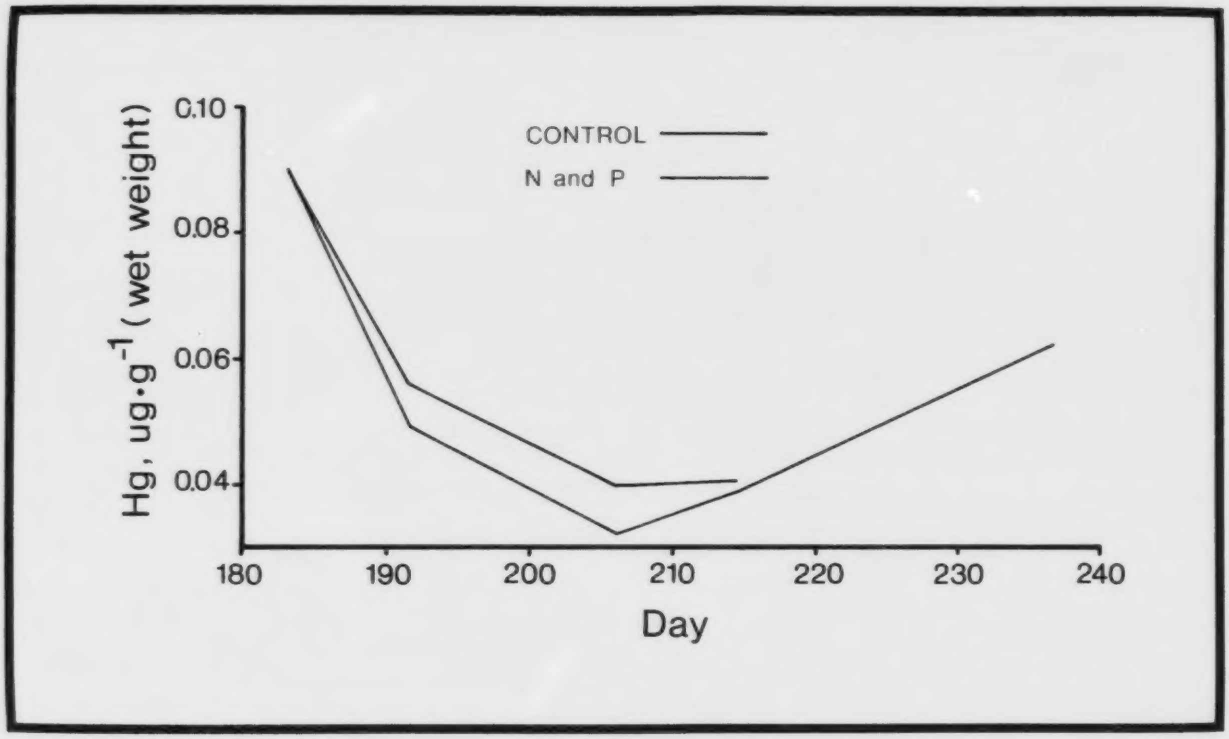
By the end of August (the completion time for most of the limnocorral experiments), the total mercury concentrations in yellow perch were indistinguishable between the three spiked limnocorrals. This was also the case between the two limnocorrals that were treated with the moss-peat material (6, 15), (Fig. 3.12). However, by the end of September, the limnocorral with the highest level of added mercury showed the highest concentration of mercury in yellow perch muscle. Similarly, the

Inundation (1980)
Northern Flood Committee

Non-flooded Granville Lake
Agassiz North Association

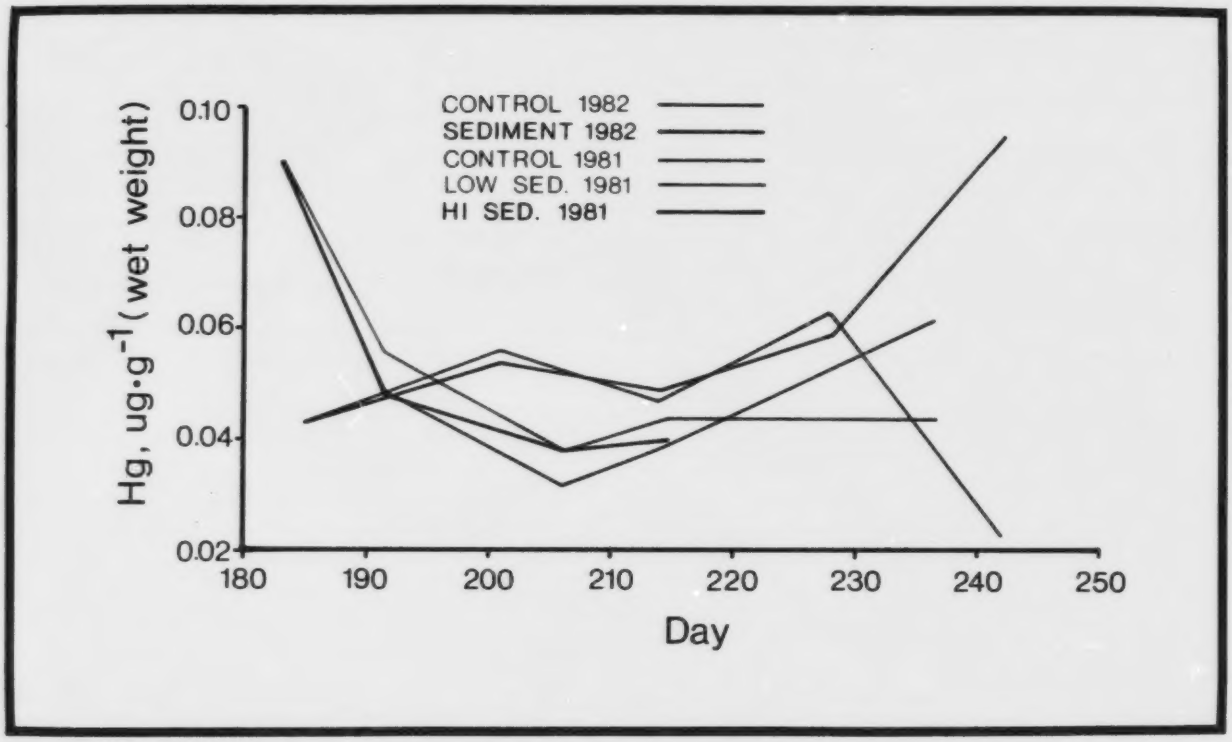
Southern Indian Lake
Department of Fisheries
and Oceans

Fig. 3.9 **The Addition of Nitrogen and Phosphorus to Limnocorral Experiments, (Plastic Bottom), 1981**



(Some samples are of muscle tissue and others are of whole fish less gut.)

Fig. 3.10 **The Addition of Sediments to Limnocorral Experiments, 1981 and 1982**



(Some samples are of muscle tissue and others of whole fish less gut. 1981 experiments used plastic bottom limnocorrals, 1982 experiments were over natural bottom.)

Fig. 3.11 **The Addition of Moss and Lake Sediments to Limnocorral Experiments, (Natural Bottom), 1983**

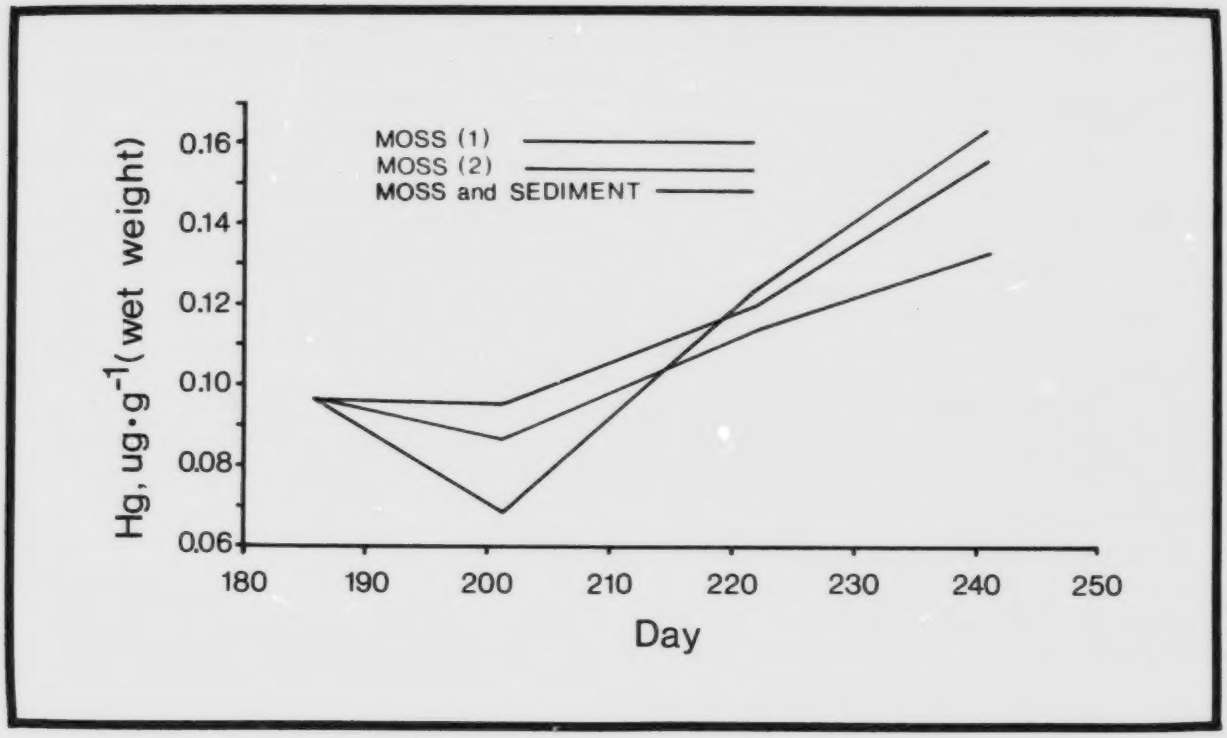
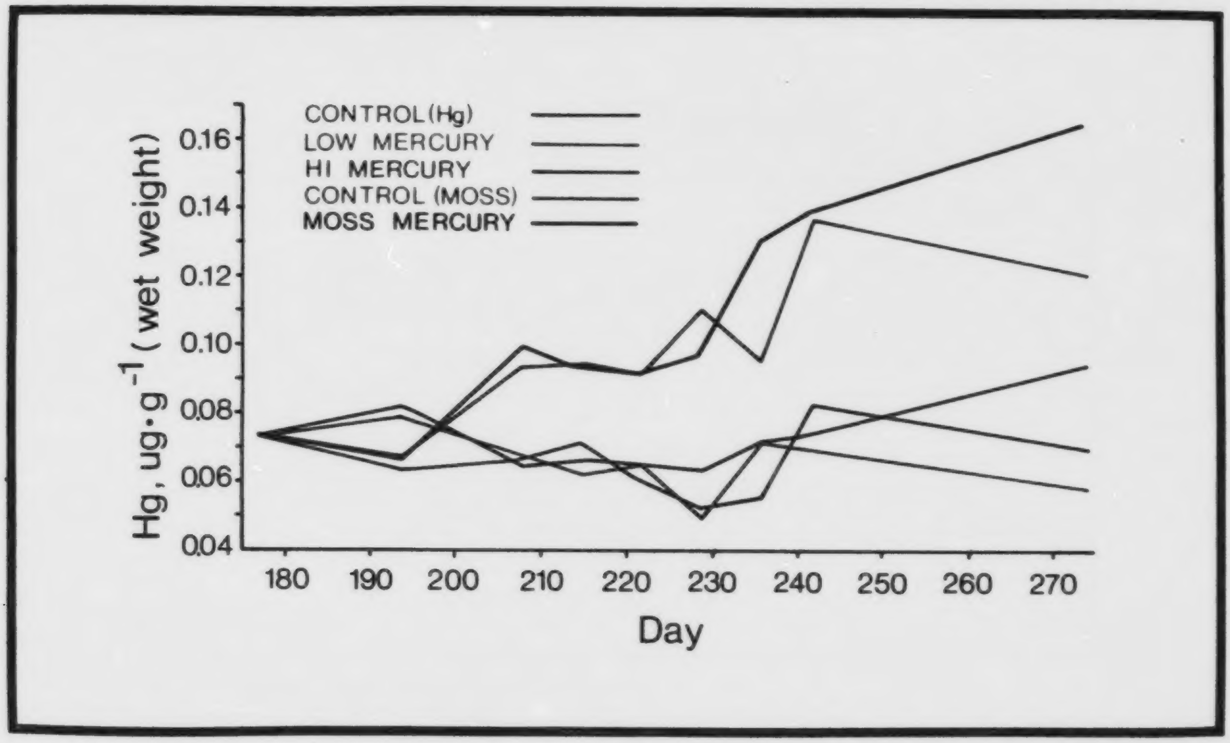


Fig. 3.12 **The Addition of Dissolved Mercury to Limocorral Experiments, (Natural Bottom), 1984**



moss-peat material limnocorral with the added mercury had a higher fish mercury concentration than did the moss-peat treated corral without the added mercury (6, 15). It was postulated that the addition of mercury resulted in the selection over three months for mercury resistant bacteria which are efficient methylators (6,26).

In summary, the addition of inorganic mercury can increase the production of methyl mercury, depending on the amount of mercury added and the length of time over which the effect is measured. This response appears to depend on its affect on the microbial community and in particular on the selection of efficient methylators within this community.

Oxygen

Laboratory experiments conducted under the Agreement on the effects of oxygen on net methyl mercury production showed that production was much higher when oxygen was excluded, provided the sulphide concentration was low (8, 25). This was generally the case in the Agreement area. Other studies have also shown that biological mercury methylation is strongly influenced by oxygen concentration, increasing dramatically when oxygen levels decline below 3 - 4 ppm (12, 22).

Reduced near-bottom oxygen concentrations were not generally found in the Agreement area (Fig. 3.13), therefore low oxygen conditions were not primarily responsible for the elevated methylation rates. However, oxygen depletion at the bottom of the Issett Lake site was sufficient to account for the much higher rate of methylation observed there than at the other sites along the diversion route (12, 22).

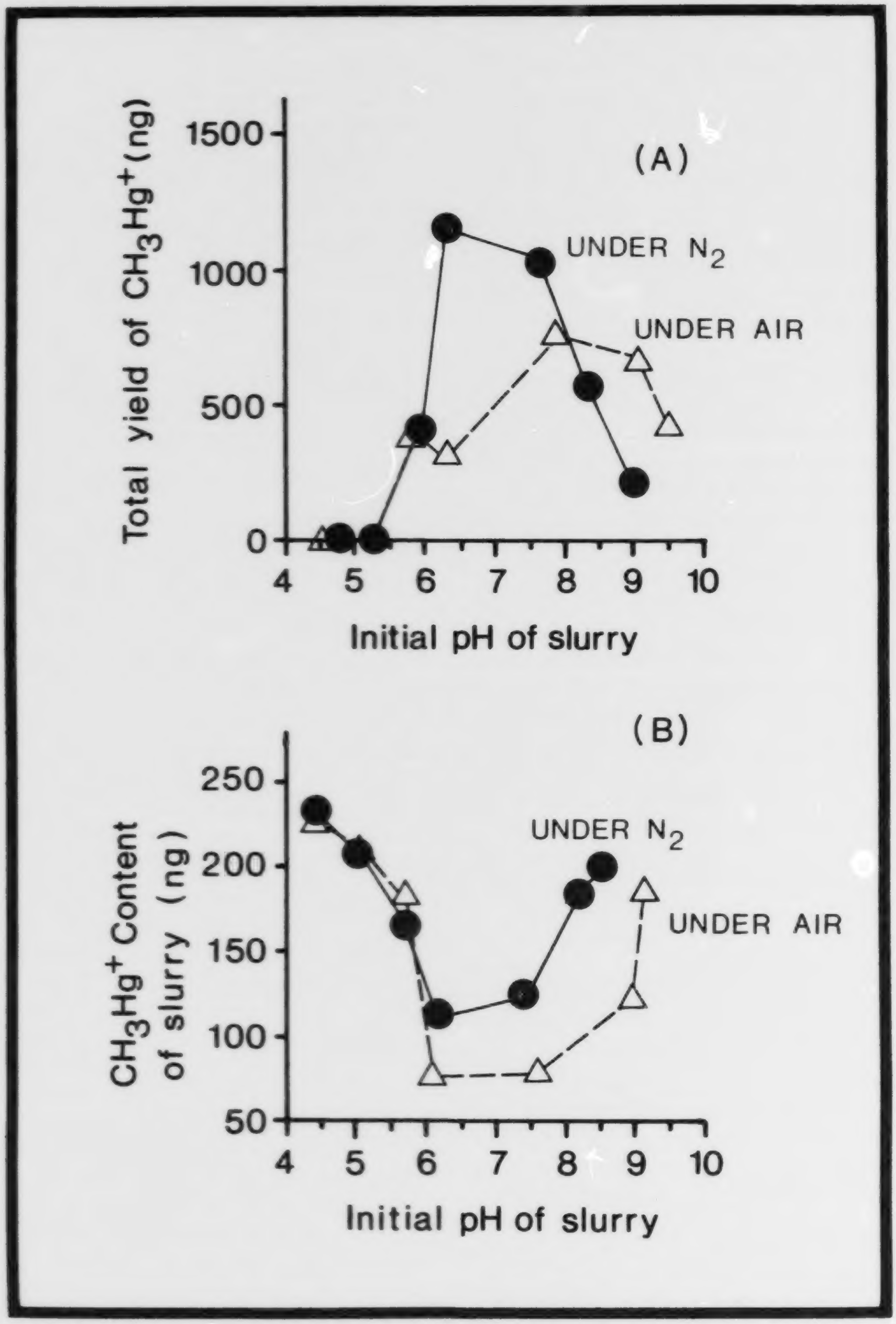
While reduced oxygen concentrations are generally not the case now, **Anoxic** conditions did occur in the bottom water of Notigi during the filling period (1974-1976) and persisted in the more poorly flushed basins even after diversion flows began to improve conditions in

Fig. 3.13 **Dissolved Oxygen Levels of Various Lakes, 1984**

TIME-WEIGHTED MEANS (ppm), 1984

Location	Inshore	Offshore
Granville Lake	9.1	8.4
East Mynarski Lake	9.4	8.7
S.I.L., Region 5	9.5	9.1
Methyl Bay	9.1	9.5
West Notigi Lake	8.6	7.1
Issett Lake	3.1	2.5
Stephens Lake	8.9	8.9
Sipiwesk Lake	9.0	8.3
Reindeer Lake	9.8	9.5

Fig. 3.14 **Effect of pH on Net Methylation**



(East Mynarski Lake sediment was enriched with moss and incubated for 7 days in the presence of a) Mercuric Chloride or b) Methyl Mercury Acetate under nitrogen.)

the main stem of the reservoir. Although now improved, Notigi Reservoir still exhibits a net consumption of oxygen as water from Southern Indian Lake passes through it. All parts of Notigi Reservoir exhibit substantially lower oxygen values than the water entering it from South Bay (9, 20).

pH

Laboratory experiments were conducted on the effects of pH on the net yield of methyl mercury. Mercuric chloride was used in some experiments, as the starting material while in others methyl mercury acetate was used. The initial pH values of the slurries ranged from 4 to 10. Fig. 3.14 shows that pH values of approximately 6.0 - 7.5 were optimal for the net production of methyl mercury.

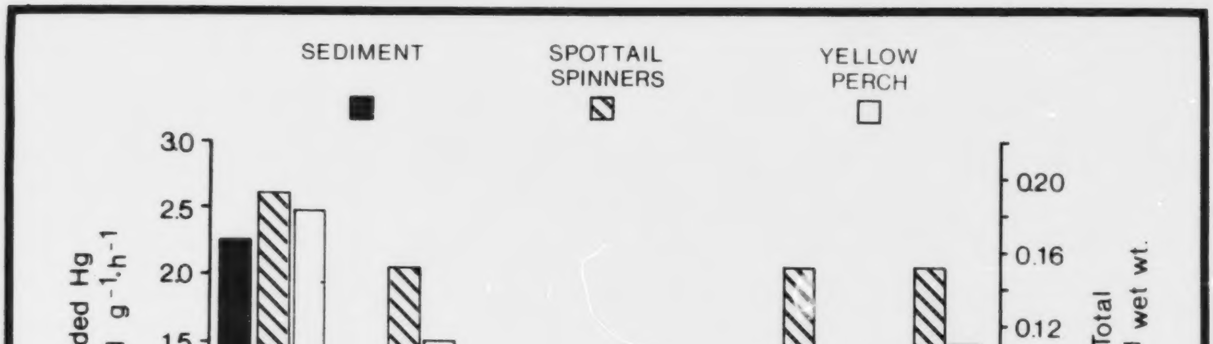
The mean pH values recorded from 12 sites in the Agreement area in 1983 were as follows: Sediment pH 6.93, bottom water pH 6.99 and surface water pH 7.36 (8, 32). Thus the pH values in the Agreement area are ideal for the net production of methyl mercury.

Temperature

Early studies under the Agreement showed a negative correlation between fish mercury concentrations and water temperatures. This result was surprising considering the seasonality of mercury methylation rates at the Methyl Bay sites in 1982 where rates were highest in mid-summer. This seasonality implies that temperature is a positive, not negative, controlling factor affecting the rate of supply of methyl mercury. Nor was there any evidence that correlations between fish mercury concentrations and water temperatures were due to an indirect effect of temperature on fish growth (3, 15).

Subsequent experiments, while not designed to specifically investigate the effect of water temperature, did not show any obvious relationship between temperature and

Fig. 3.15 **Relationship Between Mercury Methylation Rates and Fish Mercury Levels, (Survey Data), 1982**



mercury concentrations in fish (9, 14). It therefore appears that the earlier correlation may have been spurious.

METHYLATION RATES AND MERCURY IN FISH

Mercury concentrations in small fish and **Specific Rates** of mercury methylation in flooded soil were strongly positively correlated (3,12). As illustrated in Fig.3.15, West Notigi has the highest sediment methylation rates and the highest mercury concentrations in fish muscle. Fish mercury concentrations and sediment methylation rates were also comparable in West Mynarski and Area 5, Southern Indian Lake samples. Of the sites which exhibited the lowest methylation rates, only fish collected from Area 4, Southern Indian Lake had relatively high mercury levels. It is possible that the methylating bacteria at this site were sampled during a period of decline, or that the site sampled was not representative of the habitat frequented by the sampled fish (5, 10).

Unlike the 1982 mercury survey samples which were taken at sites which had been flooded for six years, the 1982 limnocorral experiments did not show any apparent relationship between the total mercury in fish muscle and the mean specific mercury methylation rates (5, 3). It was postulated that the reason why the measured methylation rates did not always reflect the total mercury levels in the fish of these enclosures was because the limnocorrals were not microbiologically in a constant state (5, 22) and the microbiota were spatially heterogeneous.

METHYLATION-DEMETHYLATION (M/D) RATIO

Since methyl mercury is both produced and degraded in the aquatic environment, the specific rates of both methylation and demethylation were measured at inshore and offshore sites in reservoirs and in control lakes. The ratio of methylation to demethylation rates (M/D ratio) was proposed as the best indicator of the net effect of the two processes, and consequently, of the potential of an aquatic system to produce high mercury concentrations in fish (12, 3). Three main observations were made from these studies.

First, specific methylation rates were generally much higher in areas of flooded terrain than in areas of non-flooded, natural lake sediments (9, 24). The inshore sites sampled in 1984 in Methyl Bay and Region 5 of Southern Indian Lake, for example, were both located over flooded terrain, while the nearby offshore sites sampled were located over non-flooded, natural lake sediment. Both the specific methylation rates and the methylation/demethylation ratios were higher for the inshore (flooded) sites than for the offshore (non-flooded) sites (Fig. 3.16). Measurements in extensively flooded reservoirs such as West Notigi, where both the inshore and offshore sites were located over flooded terrain, showed methylation rates and methylation/demethylation ratios that were as high or higher for the offshore sites as for the inshore sites.

Similarly, comparisons between reservoirs showed that specific methylation rates were markedly higher (ranging from 1.7 to 8.3 times higher) at inshore sites in reservoirs (flooded terrain) than at inshore sites in control lakes (non-flooded, natural lake sediments). In general, the methylation rates at flooded sites were strikingly similar in 1984, the only exception being the four-fold higher rates measured at the Issett Lake site (12, 9).

In contrast to the elevated methylation rates at inshore stations over flooded material, seasonal mean specific demethylation rates were either the same as or were lower in the reservoirs than in the control lakes. As a result, the M/D ratio was elevated to a greater degree than was methylation alone at all but one site (12, 10) (Fig.3.16).

Second, a distinct seasonal pattern for specific methylation rates was noted in both 1984 (12, 9) and 1985 (13, 19). This pattern, which occurred at both the inshore and offshore stations in all of the lakes and reservoirs studied, showed the highest rates occurring in August, generally during the first half of the month, with low to undetectable methylation rates at the beginning and end of the sampling period (12, 9).

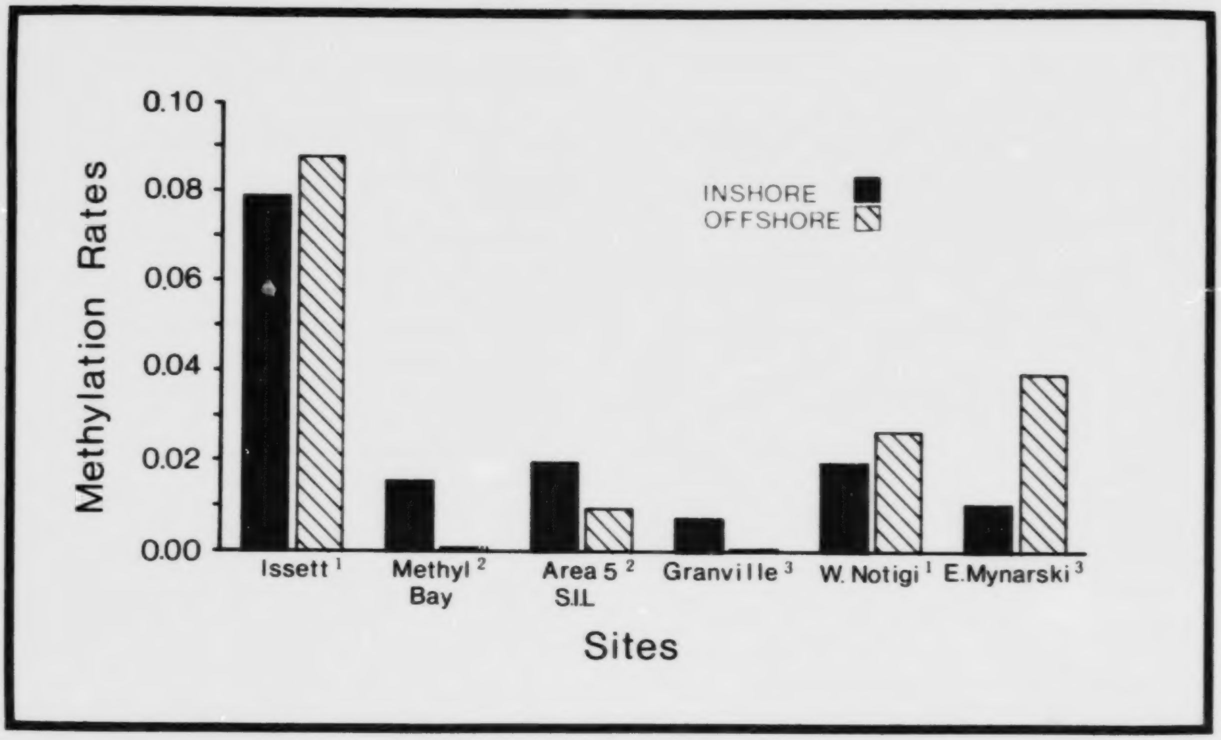
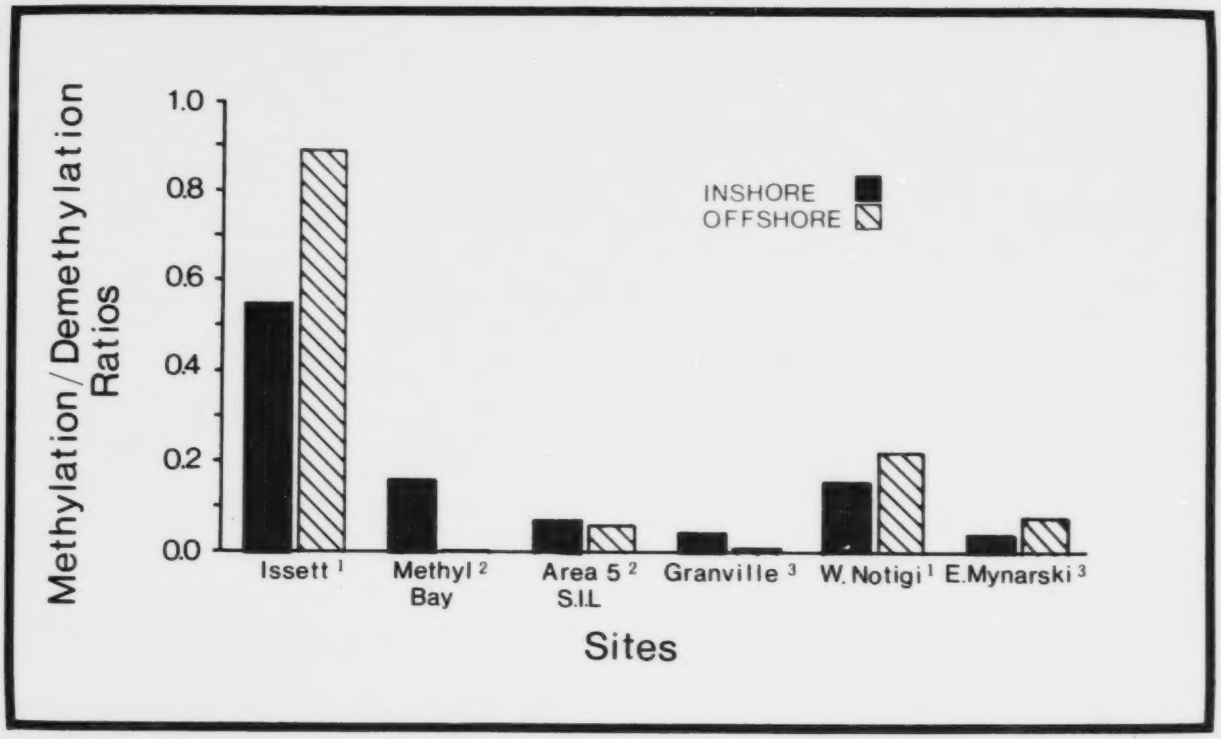


Fig. 3.16 A **Seasonal Mean Methylation Rates and Methylation/Demethylation Ratios for Inshore and Offshore Sites**

Fig. 3.16 B



(1. All sites over flooded terrain. 2. Inshore sites only over flooded terrain. 3. No site over flooded terrain.)

Evidence of a possible seasonal pattern for specific demethylation rates was less conclusive. No consistent trend was evident in 1984 (12, 20). While a seasonal trend was found in 1985, it was less distinct for one lake than for the other two lakes sampled (13, 12).

The combined effect on the methylation/demethylation ratio was to produce a strongly peaked seasonal pattern with the highest ratios occurring in late July and August in all the lakes and reservoirs examined (9, 24)(Figs. 3.17 and 3.18).

The third main observation was the lack of a relationship between specific methylation rates or M/D ratios and the mercury concentrations in sediments or water (the latter as estimated by mercury concentrations in **Zooplankton)** (9,25). However, it is interesting to note that the period of highest methylation rates and M/D ratios coincided with the period of the most rapid uptake of mercury into fish (9, 24).

The 1984 experiments showed that between August 3 (day number 215) and August 25, fish in organically treated limnocorrals achieved 70-80% of their overall change in mercury concentration, while fish in the control limnocorrals showed a slight increase during this same period. The 1984 time course of mercury uptake further substantiates the critical role of bacterial methylation in fish mercury uptake (6, 27).

Fig. 3.17 **Seasonality of Methylation/Demethylation Ratios of Inshore Sites, 1984**

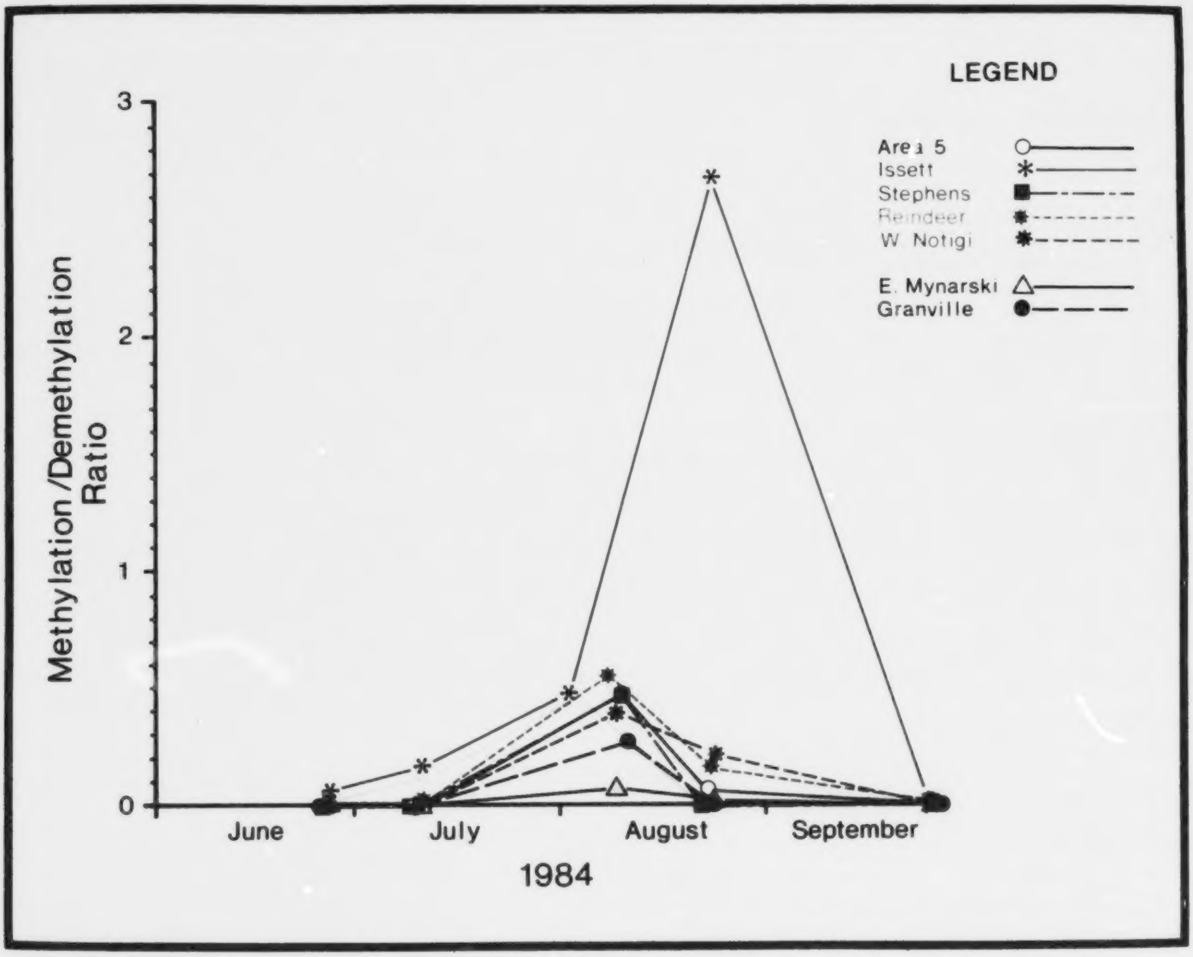
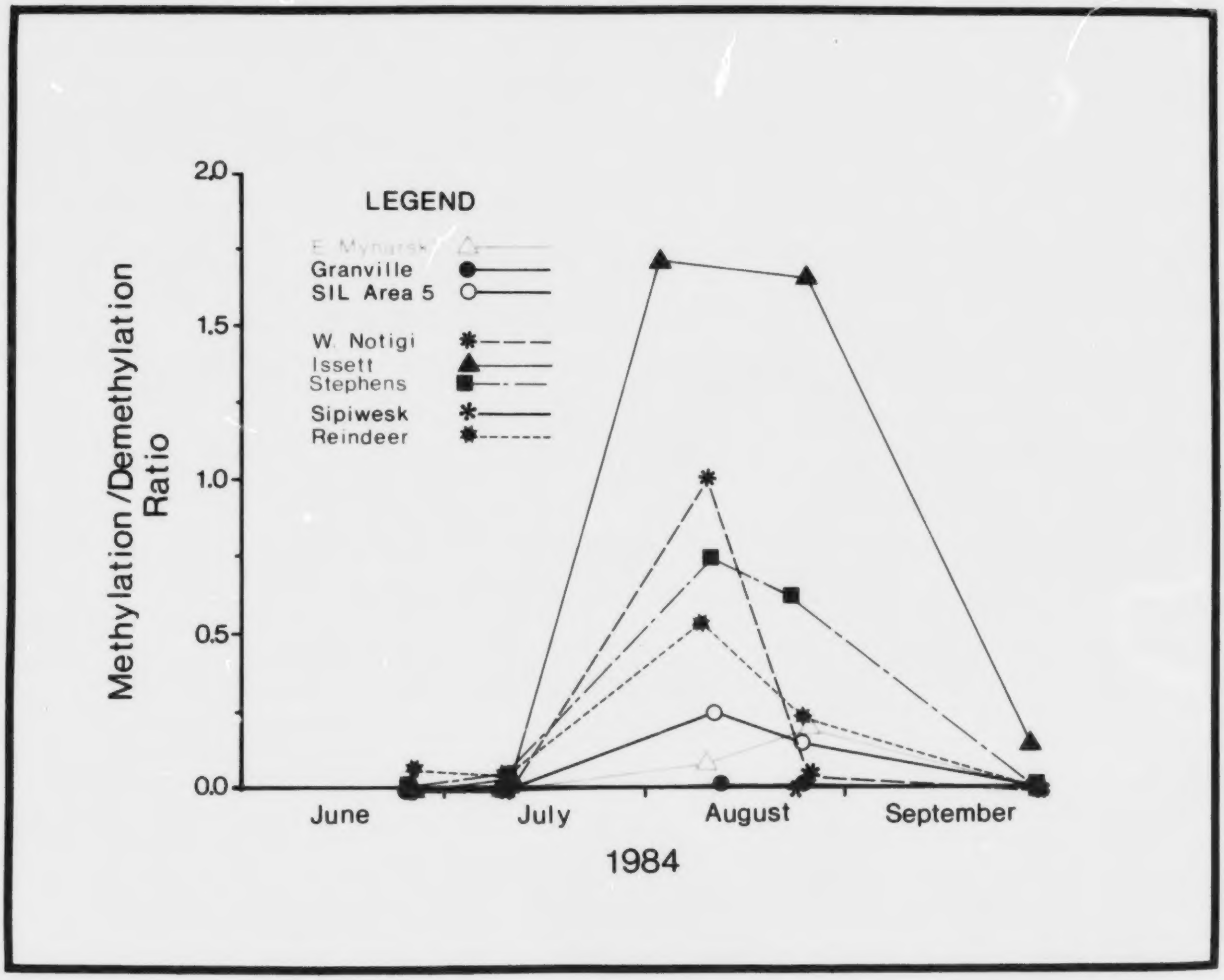


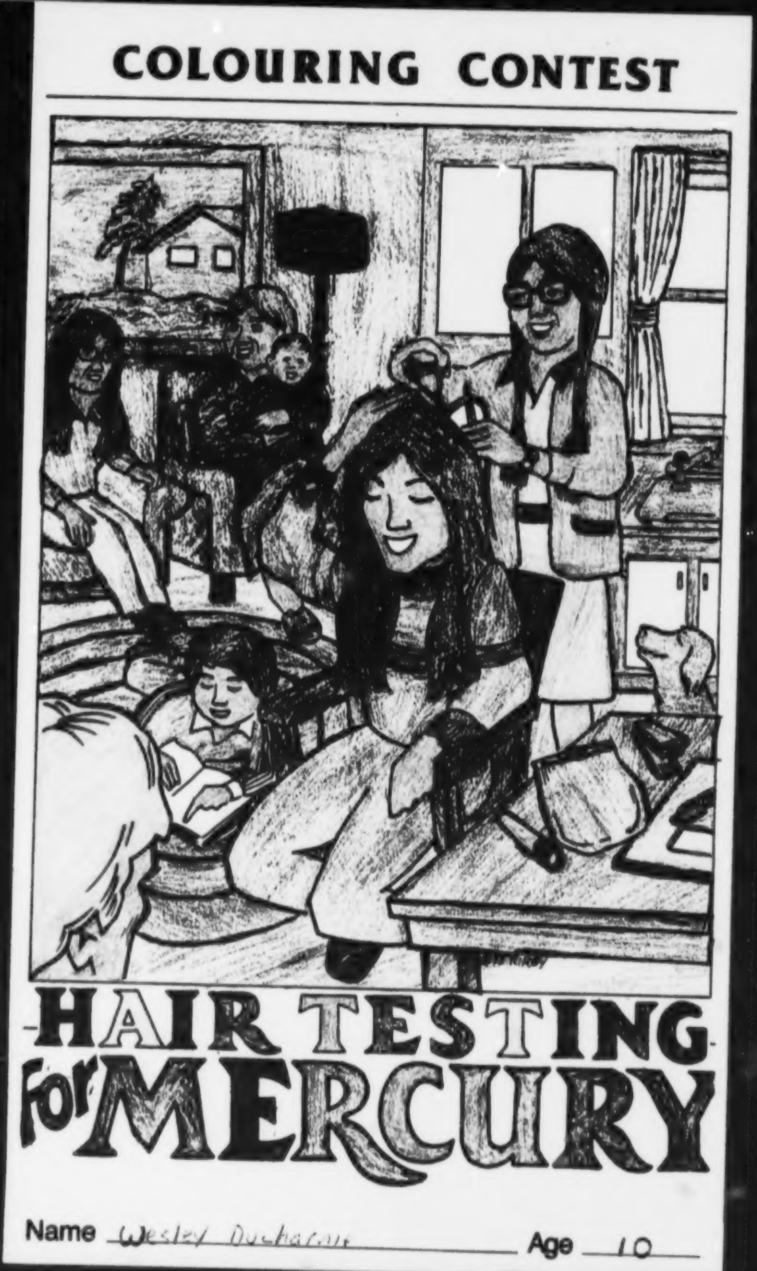
Fig. 3.18 **Seasonality of Methylation/Demethylation Ratios of Offshore Sites, 1984**



MERCURY UPTAKE BY FISH

The Agreement studies have shown the importance of the methylation/demethylation process to the methyl mercury levels in fish. However, the mechanism whereby the methyl mercury, once it has been produced by the bacterial community, accumulates in fish (either directly from water through the gills and/or from food) remains a matter of scientific debate.

The objective of one of the Agreement's studies was to try to determine the mechanism of mercury transfer between the bottom-sediment reservoir and the fish population. This study focussed on the **Benthic** community and in particular on the role of sediments in the transmission of mercury from this community. Unfortunately the results of this study were inconclusive.


COLOURING CONTEST
HAIR TESTING
for MERCURY

Name Wesley Ducharme Age 10

CHAPTER 4

MERCURY LEVELS IN PEOPLE AND ANIMALS

- INTRODUCTION
- MERCURY TOXICOLOGY
- TESTING PROGRAM
- PROGRAM RESULTS
- PUBLIC INFORMATION
- MERCURY LEVELS IN MAMMALS

INTRODUCTION

Elevated mercury levels in fish are of concern because of the potential for elevated mercury levels in fish eaters, most especially in people. While the emphasis in this chapter is on the mercury testing program in people, the chapter concludes with a brief presentation on the mercury testing program in aquatic carnivores.

The mercury testing program in people, a national program conducted by Health and Welfare Canada, was intended to identify individuals with elevated mercury levels. In order to put the results of this program into perspective, some background information on the toxicology of mercury and the specifics of the testing program must first be given.

MERCURY TOXICOLOGY

Methyl mercury as a significant environmental contaminant only came to world attention in the 1960's, following major epidemics of poisoning in Japan. Other outbreaks of methyl mercury poisoning have subsequently been reported from other parts of the world, including the largest and best documented outbreak which took place in Iraq in 1971-72. Scientific studies of the available data from such outbreaks have been conducted, including one by the World Health Organization expert group (1976). This latter study concluded that the signs and symptoms of methyl mercury poisoning would appear in the most sensitive individual adults at blood concentrations between 200 and 500 parts per billion (ppb), that prenatal effects may occur at lower levels, and that a safety factor of 10 should be used to define safe levels in man.

It is very difficult to distinguish the signs and symptoms of methyl mercury poisoning, except at very high levels of exposure, because they are easily attributable to a number of other causes. The signs and symptoms in adults include disturbance of superficial and deep sensation, constriction of the visual field, lack of coordination,and impairment of speech, hearing and gait. When exposure to methyl mercury occurs during the fetal stage, the signs and symptoms may include delayed motor and intellectual development later in life.

Methyl mercury poisoning has several important features including the fact that damage is almost exclusively limited to the nervous system, especially the central nervous system. Observations on both human and animal subjects indicate that the developing central nervous system of the fetus is more sensitive to damage from methyl mercury than is the adult nervous system. For example, studies in Japan found that some mothers who were slightly poisoned gave birth to infants with severe cerebral palsy.

Other features of methyl mercury poisoning include a long latency period (usually lasting several months) and the irreversibility of damage in severe cases as a result of the destruction of neuronal cells. The rate of intake of methyl mercury relative to its rate of elimination will determine the rate of accumulation of methyl mercury and hence the possibility of methyl mercury poisoning. Studies have shown that in man an average of 70 days is needed to eliminate half of the methyl mercury in his body, assuming no additional intake of mercury.

In summary, it is difficult to diagnose mild cases of methyl mercury poisoning, while the consequences of severe cases of poisoning are serious. There is still no specific medical treatment for the effects of methyl mercury poisoning. As a result, an emphasis has been placed on preventative measures and early detection. The mercury testing program is an important part of this process.

TESTING PROGRAM

Growing recognition of the potential health effects of environmental mercury led to the establishment of a national mercury testing program by Health and Welfare Canada in 1975. The objective of this program, which began in Manitoba in 1976, is to determine the mercury levels in native people and to provide advice on how to lower their mercury levels.

Special emphasis is given to testing heavy fish eaters, women of child bearing age, and children younger than 4 years of age. Health and Welfare Canada also identifies certain areas and communities as being 'at risk' if individuals have been found to have mercury levels in excess of 100 ppb in their blood or if the lakes fished by the community have had mercury levels in fish in excess of 0.2 ppm. The level of 0.2 ppm is used rather than the 0.5 ppm Canadian limit because native people on reserves are likely to consume large quantities of fish.

Testing for mercury levels is almost always done by analysis of hair samples. The procedure is accurate, fast, and painless, and can provide a historical record of the person's mercury levels, depending on the length of the hair. Hair concentrations of methyl mercury are proportional to the blood concentrations at the time of formation of the hair strand (Fig. 4.1). Once incorporated into the hair strand, the concentration of mercury in the hair remains unchanged.

Health and Welfare Canada offered extensive community sampling programs in six native communities in or near the Agreement area, South Indian Lake, Nelson House, Split Lake, York Landing, Cross Lake and Norway House. The results of the programs in South Indian Lake and Nelson House are presented in the next section, with comparisons to other native communities.

Fig. 4.1 **Relationship between Mercury Levels in Human Hair and Blood**

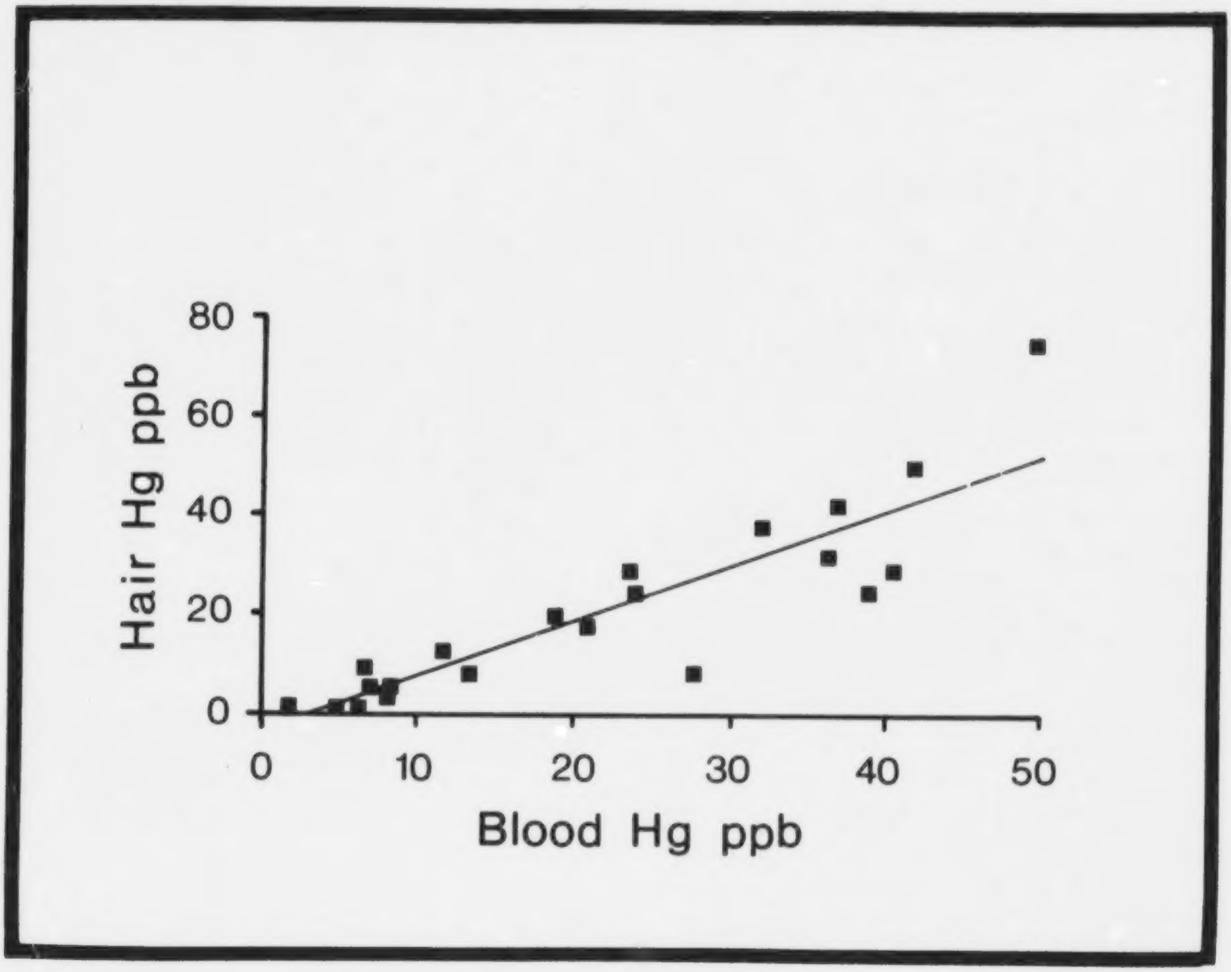


Fig. 4.2 **Mercury Levels of Residents of South Indian Lake, 1977-1985**

Date	Samples number	0 - 19ppb number	20 - 99ppb number	>99ppb number	Samples % population	0 - 19ppb % samples	20 - 99ppb % samples	>99ppb % samples
1977	37	29	8	0	5.6%	78.4%	21.6%	0.0%
1978	12	9	3	0	1.8%	75.0%	25.0%	0.0%
1979	100	26	68	6	15.3%	26.0%	68.0%	6.0%
1980	1	1	0	0	0.2%	100.0%	0.0%	0.0%
1981	165	102	63	0	25.2%	61.8%	38.2%	0.0%
1982	389	274	115	0	59.4%	70.4%	29.6%	0.0%
1983	59	55	4	0	9.0%	93.2%	6.8%	0.0%
1984	105	70	35	0	16.0%	66.7%	33.3%	0.0%
1985	56	31	25	0	8.5%	55.4%	44.6%	0.0%
	Total	Total	Total	Total		Overall	Overall	Overall
	924	597	321	6		64.6%	34.7%	0.6%

PROGRAM RESULTS

South Indian Lake

The results of the mercury testing program in South Indian Lake are presented in Fig. 4.2. These data show that for every year, except 1979, the majority of the people tested had mercury levels in the normal range of 0-20 ppb. In 1979, the majority of the people tested had mercury levels in the range of 20-99 ppb, while 6 samples exceeded 99 ppb. The percentage of samples over 20 ppb were (statistically) significantly higher in 1979 than in 1976. The years after 1979 were significantly lower but no trend with time was seen after 1983.

Fig. 4.3 shows the distribution of mercury levels in all females tested (335) in South Indian Lake of child bearing age at the time of sampling. While the majority of women tested had blood levels within the normal range of 0-20 ppb (the median blood level was 11.3 ppb), there were a number of women of child bearing age whose blood levels exceeded the generally recognized safe limit of 20 ppb, including one with a value of 125 ppb.

Mercury analysis of umbilical cord blood samples was undertaken from 1982 to 1985. Of the sixteen samples taken, most were within the normal range (median value was 8 ppb). However there were two values over 20 ppb (40 ppb and 52 ppb).

In summary, the risk to the adult members of South Indian Lake would appear to be negligible. There is some concern that a number of women of child bearing age had mercury levels which exceeded 20 ppb and that one value of 125 ppb was found (18, 19).

Nelson House

The results of the mercury testing program in Nelson House are presented in Fig. 4.4. These data show that for every year, except 1978, the majority of the people tested had mercury levels in the normal range of 0-20 ppb. In 1978, the majority of the people tested had mercury levels in the range of 20-99 ppb. There was one value of 108 ppb. The only other sample above 99 ppb recorded in Nelson House occurred in 1984. The percentage of samples above 20 ppb were higher in both 1978 and 1979 than in 1976. Levels in later years were low. Samples collected from 1983 to 1986 were significantly lower than the 1978-79 levels.

Fig. 4.5 shows the distribution of mercury levels in all females tested (254) in Nelson House of child bearing age at the time of sampling. While the majority of women tested had blood levels within the normal range of 0-20

Fig. 4.3 **Distribution of Mercury Levels in Females of Child-Bearing Age from South Indian Lake**

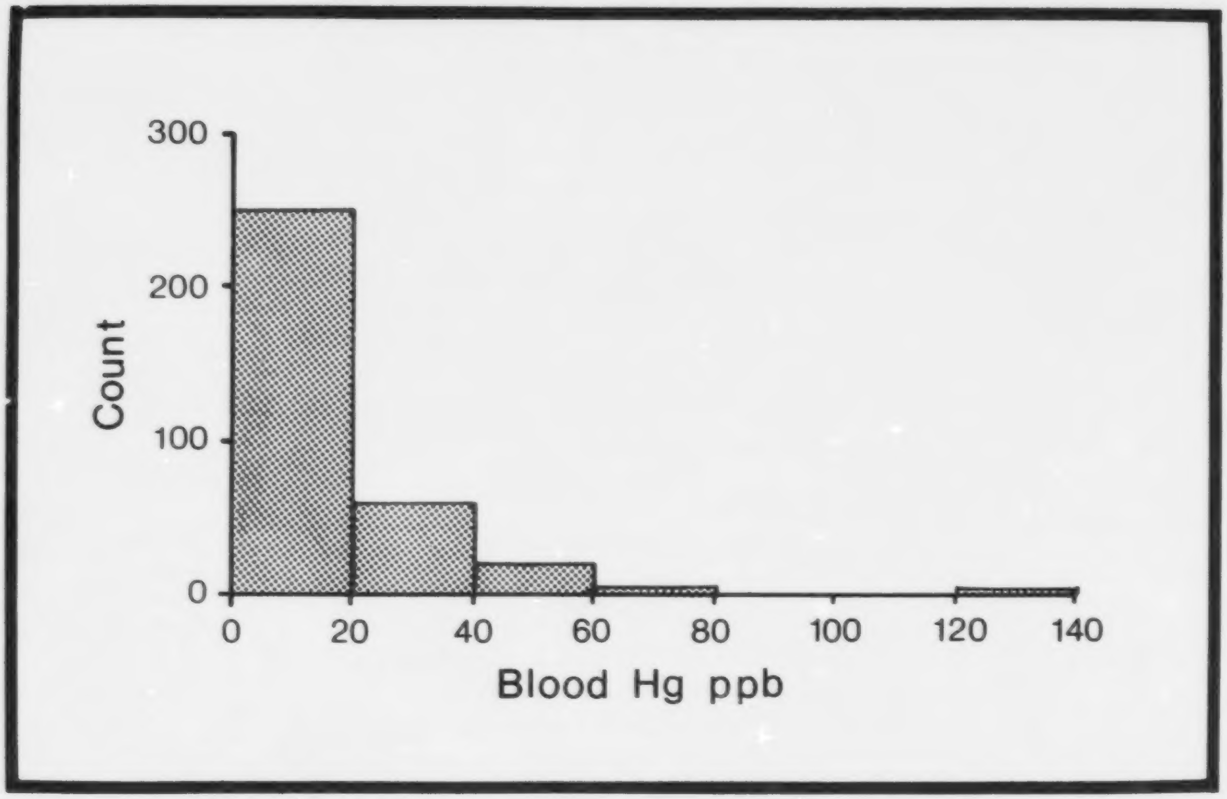
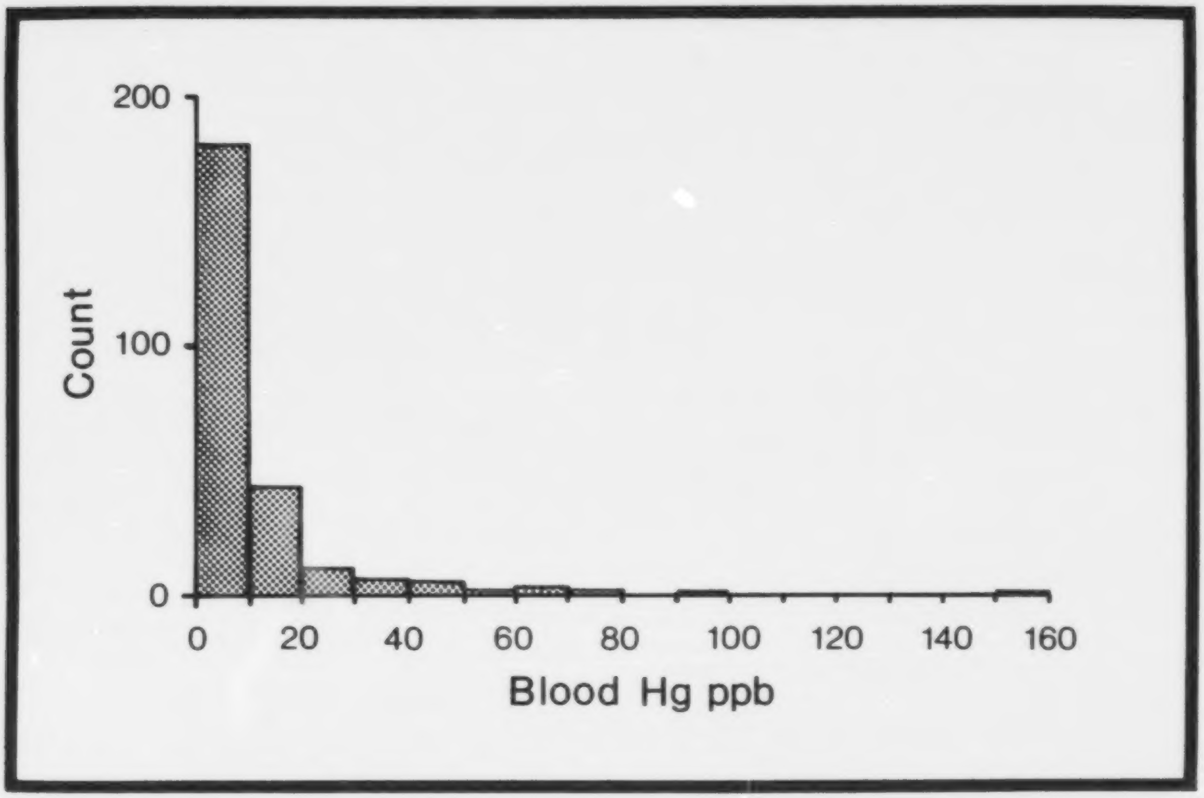


Fig. 4.4 **Mercury Levels of Residents of Nelson House, 1976-1985**

Date	Samples number	0 - 19ppb number	20 - 99ppb number	>99ppb number	Samples % population	0 - 19ppb % samples	20 - 99ppb % samples	>99ppb % samples
1976	4	1	3	0	0.3%	25.0%	75.0%	0.0%
1977	59	57	2	0	4.4%	96.6%	3.4%	0.0%
1978	102	63	38	1	7.7%	61.8%	37.3%	1.0%
1979	130	71	59	0	9.8%	54.6%	45.4%	0.0%
1980	0	0	0	0	0.0%			
1981	0	0	0	0	0.0%			
1982	56	50	6	0	4.2%	89.3%	10.7%	0.0%
1983	24	24	0	0	1.8%	100.0%	0.0%	0.0%
1984	221	181	39	1	16.6%	81.9%	17.6%	0.5%
1985	121	93	28	0	9.1%	76.9%	23.1%	0.0%
	Total	Total	Total	Total		Overall	Overall	Overall
	717	540	175	2		75.3%	24.4%	0.3%

Fig. 4.5 **Distribution of Mercury Levels in Females of Child-Bearing Age from Nelson House**



ppb (the median value was 6 ppb), there were a number of women of child bearing age whose blood levels exceeded the generally recognized safe limit of 20 ppb, including one with a value of 150 ppb.

Only three umbilical cord blood samples were analyzed for mercury (in 1982 and 1983). All were below 4 ppb.

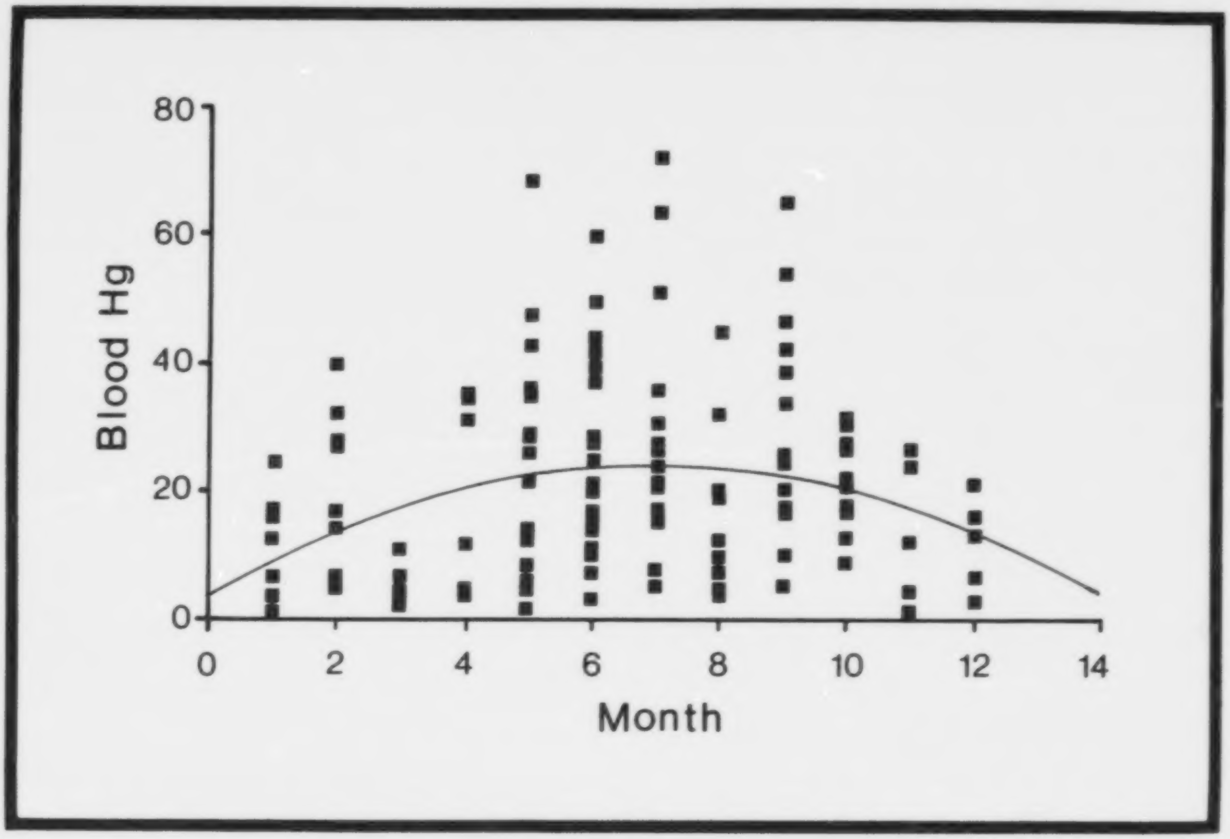
In summary, the risk to adult members of Nelson House would also appear to be negligible. There is some concern that a number of women of child bearing age had mercury levels which exceeded 20 ppb and that one value of 150 ppb was found (18, 29).

Comparison to Other Native Communities

Special consideration was given to the analysis of the mercury levels of females, particularly those of child bearing age, because of the sensitivity of the fetus to methyl mercury. When the results of the mercury samples from all females from the six northern Manitoba communities were examined, a seasonal trend in mercury concentrations was noted. The highest values were seen in the summer and the lowest in mid-winter (Fig. 4.6.) This seasonal pattern, for both males and females, has also been observed in other Canadian native communities (18, 7).

This seasonal pattern of exposure to methyl mercury is almost unique. This uniqueness has complicated assessment of the Canadian situation, but it may also have limited the severity of the health effects. For example, comparison of the English-Wabigoon River system of northwestern Ontario with Minamata Bay, Japan shows that while the range of mercury levels in the fish were similar, the mercury levels of the native people along the English-Wabigon River system were an

Fig. 4.6 **Seasonality of Mercury Levels in Females of Six Northern Manitoba Communities**



order of magnitude lower than in Japan where a more sustained exposure to methyl mercury occurred (18, 9).

Within the six native communities in or near the Agreement area, South Indian Lake followed by Nelson House have the highest reported mercury levels, whether viewed as the percentage above 20 ppb or the percent above 99 ppb. York Landing and Split Lake have intermediate levels significantly below those of Nelson House and South Indian Lake. Cross Lake and Norway House had the lowest reported values significantly below York Landing and Split Lake. A number of factors can account for these differences between communities, including differences between communities, including differences in the percentage of each community sampled, differences in the number of repeat samples, non-random sampling, seasonal changes, differences in fish consumption patterns, and differences in the mercury levels of the fish consumed (18, 2).

Five communities in Ontario and Quebec with some of the highest reported mercury levels in those provinces, if not in all of Canada, were selected for comparison with the six northern Manitoba communities (18, 11). These five communities differ sharply from the six Manitoba communities in the percentage of samples above 99 ppb (Fig. 4.7). In terms of the percentage of samples above 20 ppb, South Indian Lake falls within the range of the five Quebec and Ontario communities, while Nelson House is at the bottom of this range.

In summary, the mercury levels of the people tested in the communities of South Indian Lake and Nelson House are significantly higher than

Fig. 4.7 **Comparison of Mercury Levels in Eleven Canadian Native Communities, 1971-1986**

Community	Population	Samples number	% Population Sampled	% of Samples >20 ppb	% of Samples >99 ppb
			MANITOBA		
SOUTH INDIAN LAKE	655	924	89.8%	35.3%	0.6%
NELSON HOUSE	1331	717	33.4%	24.7%	0.3%
YORK LANDING	425	369	53.2%	17.6%	0.0%
SPLIT LAKE	1037	870	60.2%	16.7%	0.0%
CROSS LAKE	2079	988	39.9%	8.0%	0.0%
NORWAY HOUSE	2335	800	25.4%	6.0%	0.0%
			ONTARIO AND QUEBEC		
MISTASSINI, QUE.	1733	6580	99	44.0%	4.0%
Waswanipi, Que.	742	2180	100	40.6%	5.1%
Grassy Narrows, Ont.	477	2578	100	32.6%	6.3%
Chisasi, Que.	1811	2040	90 - 100	29.5%	2.0%
Whitedog, Ont.	630	2759	85 - 100	24.%0	2.5%

those tested from four other northern Manitoba communities. In terms of the percentage of samples above 99 ppb, South Indian Lake and Nelson House are considerably lower than other Canadian native communities that are also considered to be 'at risk'.

PUBLIC INFORMATION

An important objective of the mercury testing program in the six communities in or near the Agreement area is ensuring that as many individuals as possible are tested. The Agreement office, working in conjunction with Health and Welfare Canada, used this objective as the cornerstone of the Agreement's public information program to South Indian Lake and Nelson House, the communities of most concern.

A number of public information activities were undertaken with the intent of increasing the awareness and understanding of mercury by the residents of the Agreement Area. These activities included public meetings on mercury in South Indian Lake and Nelson House, newsletter articles, community school programs, meetings with special interest groups such as area trappers, and the production and distribution of English and Cree language videotapes on mercury.

A whitefish recipe contest was held for adults (to draw attention to their lower mercury content) and a colouring contest for children (with a mercury-in-hair sampling scene as the subject of the colouring poster). The latter contest was particularly successful with almost 300 entries received.

With the conclusion of the Agreement, the public information activities on mercury which were funded under the Agreement have also concluded. However, the duration and extent of the testing program conducted by Health and Welfare Canada will continue to be evaluated yearly by them. The provision of public information on the health aspects of mercury, particularly, in the form of appropriate dietary advise will continue to be an important part of the Health and Welfare Canada program.

MERCURY LEVELS IN MAMMALS

Two aquatic carnivores, mink and otter, were monitored to determine their mercury levels since it is known that they accumulate mercury concentrations in their tissues in excess of those of their prey (15,2). The trapping of these animals for fur, which

Fig. 4.8 **Mercury Levels in Liver Tissue of Manitoba Mink, 1982-1985**

A. Individual Areas

Mercury Levels in freeze-dried tissue, (µg/g). Median (range)

AREA	YEAR			OVERALL YEARS
	1982/83	1983/84	1984/85	
Southern Indian Lake	---	10.90 (2.09-74.7)	7.18 (3.97-43.0)	9.92
Nelson House	---	15.90 (5.14-42.4)	7.85 (0.52-55.6)	8.61
Split Lake	8.68 (1.40-18.5)	8.10 (2.32-26.3)	3.63 (0.37-62.1)	4.57
Pikwitonei	---	---	2.60 (0.30-21.7)	2.60

B. Area Totals

AREA	YEAR			OVERALL YEARS
	1982/83	1983/84	1984/85	
Agreement Area[1]	8.68	12.05	5.48	6.62
Southern Manitoba	2.73 (0.29-29.4)	3.71 (0.36-19.5)	---	3.13

[1] Agreement area = Southern Indian Lake, Nelson House, and Split Lake Registered Trapline Sections.

provides a source of income for some area residents, made it especially important to know whether these animal populations were in danger.

From 1981-85, 403 mink carcasses were obtained throughout Manitoba for mercury analysis, of which 147 were from the Agreement area. From 1982-85, 55 otter carcasses from Manitoba were obtained, of which 34 were from the Agreement area. Mercury concentrations were determined for brain, liver and kidney tissues for all animals. The results of the analyses of the liver tissues of mink and otter are shown in Figs. 4.8 and 4.9 respectively.

Statistical analysis of the mercury concentrations in mink for all 3 tissues showed generally the same trend. Of the 3 Agreement areas sampled, the values for the Southern Indian Lake and Nelson House areas were not significantly different, while the Split Lake area had the lowest values. The Agreement area samples had higher mercury concentrations than did the other Manitoba samples analyzed (15, 11).

Statistical analysis to determine changes from year to year showed that concentrations in the Split Lake area had significantly decreased from 1984 to 1985. While those in the Southern Indian Lake area also appeared to decrease, analysis did not show the change to be statistically significant. In the Nelson House area the decrease was statistically significant only for liver tissues (15, 9).

The trends in the otter mercury concentrations appear to be similar to those observed in mink, although the differences are less definite and not always statistically significant (15, 9).

A small percentage of aquatic furbearers accumulated near-toxic concentrations of mercury in their tissues. However, there does not appear to be any detrimental effect on their population or on the fur harvest in the area (15, 16).

Fig. 4.9 **Mercury Levels in Liver Tissue of Manitoba Otters, 1982-1985**

Mercury Levels in freeze-dried tissue, (µg/g). Median (range)

AREA	YEAR			OVERALL YEARS
	1982/83	1983/84	1984/85	
Southern Indian Lake	---	17.10 (5.2-17.3)	11.60 (N = 1)	14.35
Nelson House	---	11.67 (8.33-15.0)	---	11.67
Split Lake	3.03 (N = 1)	12.10 (3.28-43.0)	4.32 (2.01-17.8)	4.87
Pikwitonei	---	---	3.51 (1.26-14.5)	3.51
Agreement Area[1]	3.03	13.55	5.00	9.04

[1] Agreement area = Southern Indian Lake, Nelson House, and Split Lake Registered Trapline Sections.

Fig. 4.10 **Mercury Concentrations in the Hides of Small Mammals from Agreement Area, 1984**

A. DEER MICE (Peromyscus maniculatus)

Mercury Levels in Freeze-dried Tissue (μg/g)

LOCATION	WATER EDGE		INLAND	
	N	Median	N	Median
Notigi Lake	19	3.42	7	1.94
Threepoint Lake	5	1.10	4	0.74
Odei River	10	0.11	14	0.06

B. RED BACKED VOLES (Clethrionomys gapperi)

LOCATION	WATER EDGE		INLAND	
	N	Median	N	Median
Notigi Lake	3	0.24	6	0.15
Threepoint Lake	10	2.20	2	0.17

In addition to the spatial and temporal comparisons of mercury concentrations in mink and otter, comparisons with the mercury concentrations of northern pike were also made. The ranking of mercury concentrations in the mink and otter from different areas did not correspond to that obtained from analysis of the pike samples, nor did year-to-year changes in their mercury levels correspond (15, 13). There is no single explanation which can account for these differences.

Mercury concentrations of the deer mouse and red-backed vole were also analyzed since they are important prey not only for mink but for terrestrial predators as well, thereby providing a potential link between the aquatic and terrestrial food chain (15, 3). The mercury concentrations in the deer mice from the Notigi area were significantly higher than those from Threepoint or Odei (Fig. 4.10). Furthermore, in all three areas the animals caught along the water's edge had higher mercury concentrations than animals caught inland (15, 10).

In summary, mercury levels in mink and otter from the Agreement area are elevated in comparison to other Manitoba regions, but are below the toxic levels. Mercury concentrations in mink decreased over the last two years of sampling, but without a continuous time series of data, no conclusion can be reached regarding long-term trends.

CHAPTER 5

THE DURATION AND EXTENT OF THE MERCURY PROBLEM

- DURATION OF PROBLEM
- EXTENT OF PROBLEM
- MITIGATION MEASURES

DURATION OF THE PROBLEM

With the development of techniques to measure specific methylation and demethylation rates, it was postulated that these measurements could be used to predict the time course of the mercury problem. These measurements were examined to determine whether there was a change over time within a given reservoir, as well as to examine the balance between specific mercury methylation and demethylation rates in reservoirs of different ages.

The only reservoir for which a time record of methylation and demethylation rates exists is Methyl Bay, an isolated bay of Southern Indian Lake. Methylation rates were measured from 1982 to 1985 and demethylation rates from 1983 to 1985. There was no indication that the differential in specific methylation rates between flooded and offshore stations in Methyl Bay diminished during this time period (Fig. 5.1). Year to year variation was of the same order as the variation among the stations in a given year (13, 16). This also appeared to be true for the methylation/demethylation (M/D) ratio. The persistance of elevated methylation rates ten years after reservoir formation would suggest that the mercury levels in the fish will continue to be elevated for some time.

In 1984, mercury methylation and demethylation rates were measured in five reservoirs, of varying ages, and two control lakes in or near the Agreement area (Fig. 5.2). All but one reservoir showed methylation rates which were high relative to the natural lakes. The one exception, Sipiwesk Lake, was sampled during a low water stage which made it impossible to sample over flooded terrain (12, 16).

The fact that methylation rates and the M/D ratios were as high in flooded areas of 14 year old Stephens Lake and 42 year old Reindeer Lake as in eight year Southern Indian Lake suggests that enhanced methylation rates and elevated M/D ratios may persist in the flooded areas of reservoirs for a very long time. Decades rather than years may be the appropriate time scale for northern reservoirs for the return to pre-impoundment M/D ratios and mercury levels in fish.

In 1985, measurements of methylation and demethylation rates were made in Granville Lake (a control lake), Southern Indian Lake, and Cedar Lake. Cedar Lake, located approximately 200 km south of the diversion area, was flooded in 1964 and 1965 to provide storage for the Grand Rapids Hydroelectric Generating Station. The specific methylation rates and M/D ratio in Cedar Lake were found to be comparable to those of Granville Lake (Fig. 5.3). The low methylating activity is consistent with the low fish mercury levels currently found in Cedar Lake (13,13).

The relatively short duration of the mercury problem here contrasts with the prediction for the lakes along the diversion route. This may be due to the fact that the impoundment of Cedar Lake flooded a very different area from that flooded by the diversion lakes and thus may not be an appropriate model for the recovery of other northern Manitoba reservoirs (9, 25).

Cedar Lake flooded an extensive area of the sandy delta of the Saskatchewan River which had predominately marsh vegetation. The other reservoirs studied were in boreal forest with peaty organic soils. The dynamics of organic degradation are likely quite different for these different vegetation and soil regimes (9, 26).

EXTENT OF PROBLEM

In the last 10 years there has been a growing awareness of an association between reservoir impoundment and high mercury levels in fish. Recently, two other studies have been completed which document both pre- and post-impoundment levels of mercury in fish, one in northern Quebec, Canada and one in South Carolina, U.S.A.

A major hydroelectric development project (design capacity of 11,000 MW) is currently underway in northern Quebec, involving large scale diversions and the creation of numerous reservoirs. The area of this project bears a number of similarities to the Agreement area in terms of climate, vegetation, permafrost, fish species, and the presence of a Cree population, (a number of whom pursue a traditional lifestyle).

Measurements of mercury levels in fish were made both before and after impoundment in two reservoirs. Fig. 5.4 shows that mercury levels rose after impoundment. The piscivorous fish had the highest mercury levels, with essentially all of the walleye and pike sampled having levels after impoundment that were above the Canadian limit of 0.5 ppm. The mercury levels of a number of the whitefish sampled also rose to levels above the Canadian limit, a situation which did not occur in the Agreement area. The fact that the pre-impoundment as well as the post-impoundment mercury levels in fish in Quebec were higher than those recorded in Manitoba would suggest that the Quebec area had naturally higher mercury levels.

Pre- and post-impoundment mercury levels were also measured in fish from the multipurpose Russell Reservoir on the Savannah River in South Carolina. There too, fish mercury levels rose significantly after impoundment. For example, from September 1983 to August 1984 mean mercury levels in bass rose from 0.161 ppm to 0.544 ppm, in sunfish from 0.081 ppm to 0.395 ppm and in yellow perch from less than 0.025 ppm to 0.441 ppm.

Mercury studies on the Russell Reservoir also included measurements of mercury concentrations in soils and sediment. Analysis of mercury levels in flooded and unflooded soils showed no apparent patterns in soil mercury concentrations, however a great variation in the results between the soil subsamples was found. Analysis of sediment cores showed a decrease in mercury concentrations with

Fig. 5.1 **Specific Methylation Rates, Methyl Bay, Southern Indian Lake, 1982-1985**

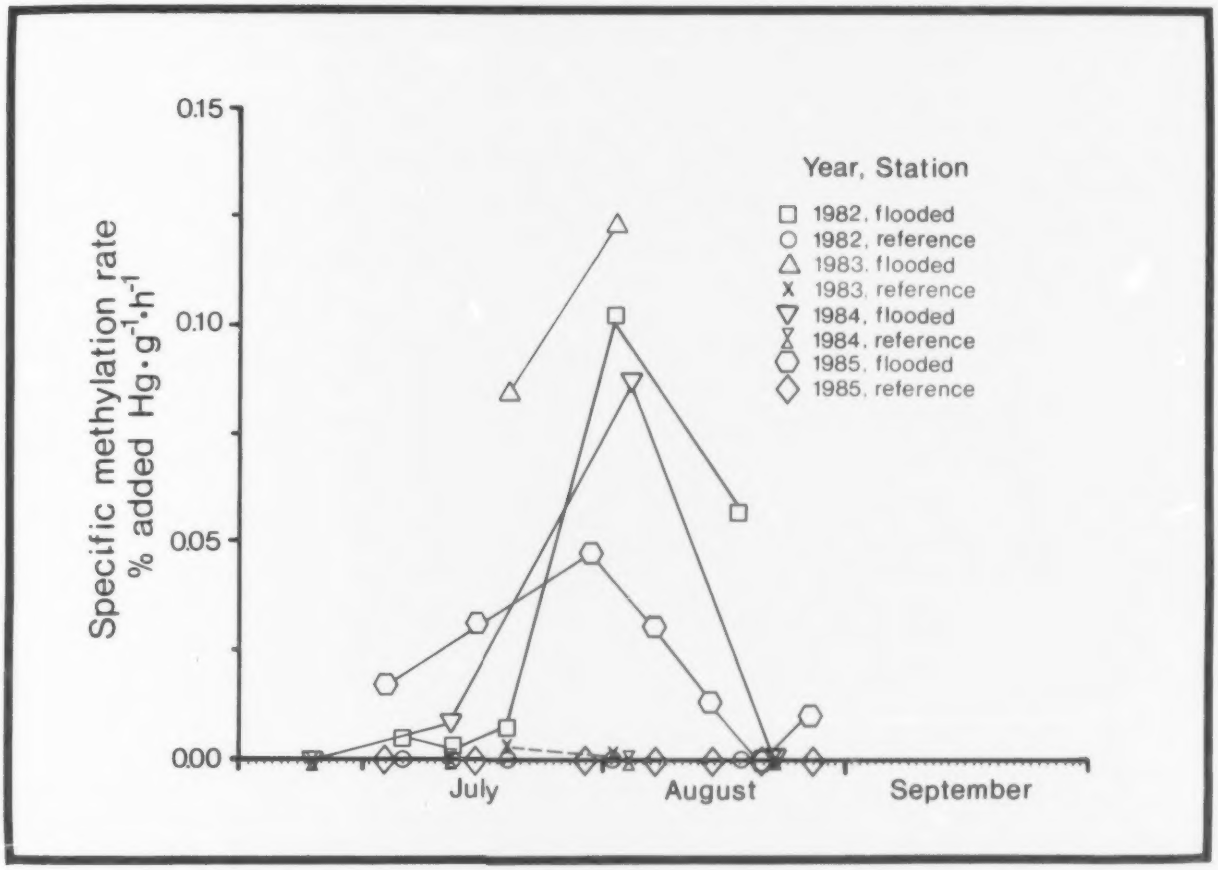


Fig. 5.2 **Seasonal Mean Specific Rates of Mercury Methylation (M) and Methyl Mercury Demethylation (D) and the M/D Ratio of Sites Surveyed in 1984**

Site	% added Hg g^{-1} h^{-1}		
	Methylation	Demethylation	M/D
Granville L.			
I-U	0.008	0.185	0.043
O-U	0.001	0.132	0.009
East Mynarski L.			
I-U	0.011	0.273	0.038
O-U	0.04	0.518	0.077
Methyl Bay (1974-76)			
I-F	0.016	0.096	0.162
O-U	<0.001	0.086	0.004
SIL Region 5 (1974-76)			
I-F	0.02	0.272	0.072
O-U	0.01	0.158	0.061
West Notigi L. (1974-76)			
I-F	0.02	0.132	0.155
O-F	0.027	0.122	0.222
Issett L. (1974-76)			
I-F	0.079	0.143	0.551
O-F	0.088	0.099	0.892
Stephens L. (1970)			
I-F	0.017	0.147	0.113
O-F	0.053	0.146	0.362
Sipiwesk L. (1960-1965)			
I-U	0.003	0.181	0.015
O-U	0.002	0.115	0.017
Reindeer L. (1939-1942)			
I-F	0.019	0.097	0.199
O-U	0.002	0.109	0.203

(   ) year of impoundment

I - inshore	F - flooded
O - offshore	U - unflooded

Fig. 5.3 **Seasonal Mean Specific Rates of Mercury Methylation (M) and the Methyl Mercury Demethylation (D) and the M/D Ratio and Sites Surveyed in 1985**

Site	% added mercury g^{-1} hr^{-1}		
	Methylation	Demethylation	M/D
Granville			
I-F	0.006	0.375	0.016
O-U	<0.001	0.865	0.006
Methyl Bay			
I-F	0.026	1.097	0.023
O-U	<0.001	0.182	0.001
Cedar Lake			
I-F	0.003	0.666	0.005
O-U	0.002	0.738	0.002

I - inshore
O - offshore

F - flooded
U - unflooded

increasing depth. These findings were similar to the findings of the Agreement studies.

An examination of the conditions most likely to result in mercury problems was one of the objectives of a study completed in 1986. This study, sponsored by the Canadian Electrical Association (CEA), concentrated on drawing together information on reservoirs from throughout Canada in an attempt to assess the risk of mercury problems in different reservoirs. Emphasis was placed on examining factors that differed among reservoirs, rather than on factors which changed as a result of flooding.

One particularly interesting comparison involved four reservoirs from the Wreck Cove development in Nova Scotia. The four reservoirs were created at approximately the same time (1978), are of the same order of magnitude in size (355-1050 ha), and are located within the same watershed. One important difference was the widely differing areas flooded relative to the surface areas of the water bodies on these sites prior to flooding.

Wreck Cove Lakes, which was the reservoir with the smallest area flooded relative to the original area of the lake, had by far the highest mercury levels in brook trout. This result could not be traced to any obvious physical or chemical differences among these reservoirs. The lack of information on pre-impoundment mercury levels in the fish made it impossible to determine the relative increase in the mercury levels in fish following impoundment for these reservoirs.

In summary, studies of other reservoirs, of which three have been briefly mentioned here, reveal the widespread nature of the mercury problem. Many factors appear to influence mercury levels which makes it impossible to predict the severity of the problem at a given reservoir, especially if there is no previous experience in the same area.

MITIGATION MEASURES

Another objective of the CEA study was to examine possible mitigation measures. A total of twelve were examined, including the addition of materials (e.g. selenium, inorganic materials), the removal of vegetation prior to

Fig. 5.4 **The Range of Mean Mercury Levels in Fish, Adjusted by Length, in La Grande 2 and Opinaca Reservoirs, Quebec**

Species & Location	Mean Mercury Concentrations (ppm)		
	Before Impoundment	1981	1982
Whitefish			
La Grande 2 (1979)	0.07-0.26	0.44-0.51	0.48-0.59
Opinaca (1980)	0.06-0.19	0.16-0.47	-
Pike			
La Grande 2	0.27-1.22	-	1.07-2.26
Walleye			
La Grande 2	0.26-1.47	0.99-1.36	1.24-2.22
Opinaca	0.26-0.55	0.83-1.73	-

(   )year of impoundment

flooding, and fish management activities. Most of the measures examined were considered to be too expensive, impractical, and/or unacceptable because of possible negative environmental impacts.

Fish management activities were considered the most feasible measures. These activities include the provision of guidelines to encourage consumption of non-predatory fish and smaller-sized fish, compensation for any losses to commercial fisheries, and the provision of alternative fishing opportunities, for example by stocking near-by water bodies where a mercury problem does not exist. The suitability of these fish management activities would depend, in part, on the expected duration of the mercury problem.

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2. Mercury Mobilization and Biomagnification Resulting from the Filling of a Piedmont Reservoir. A.R. Abernathy et al, Water Resources Research Institute, Clemson University, Clemson, South Carolina, 1985.
3. Mercury Release in Hydroelectric Reservoirs. M.C. Jones et al, Report for the Canadian Electrical Association, CEA No. 185 G399, 1986.

CHAPTER 6

CONCLUSIONS AND RECOMMENDATIONS

CONCLUSIONS

1. There is no evidence that anthropogenic sources of mercury are a direct cause of the elevated concentrations of mercury in fish of the Agreement area.

The known potential anthropogenic sources of mercury within or near the Agreement area are mining and smelting activities. Studies in northern Manitoba have indicated that the effect of mercury from mine tailings or from atmospheric fallout from smelters is limited to a small area in the immediate vicinity of these sources. Thus, it was concluded that these activities did not contribute to the elevated mercury levels in the fish of the Agreement area.

It was further concluded that the mercury in the Agreement area comes from natural sources, with the bedrock being the ultimate source. This conclusion was supported by geological studies which have shown that the volcanogenic rocks of the Agreement area contain elevated mercury concentrations with respect to both other rock types in the area and similar rocks in other parts of Canada.

2. Mercury levels in the soil, sediment and water of the Agreement area are within the normal ranges for unpolluted areas.

Virtually all of the mercury concentrations in the sediment of the Agreement area were within the range of reported levels in sediments for other unpolluted areas. Mercury concentrations in the soil from the Agreement area also compared favourably with the reported levels in soils for other unpolluted areas. However, while the mercury concentrations in the soil were low, they averaged approximately three times higher than the mercury concentrations in the sediments of the pre-impoundment lake bottoms.

Experiments were therefore undertaken to determine if there was any depletion of the mercury from flooded soils. The results showed little indication of any significant changes in the mercury concentrations in flooded soils five years after impoundment. However interpretation of the data was difficult because of the large amount of intra- and inter-site variability in soil mercury concentrations.

Measurements of mercury concentrations in water yielded results which were generally below the detection limit. Mercury concentrations in plankton were therefore used to provide a relative measure of the mercury concentrations in water. It was found that the mercury levels in the water of flooded reservoirs were similar to those of the control lakes in the Agreement area and to those of other uncontaminated systems. Thus, there was no evidence that flooding has caused an increase in the mercury concentrations in the water of any of the reservoirs studied.

3. Post-impoundment mercury levels in fish from the lakes along the diversion route are elevated in comparison with Canadian guidelines, with pre-impoundment levels and with mercury levels in fish from other non-flooded lakes within the Agreement area. Mercury levels in fish also varied according to species and size, as well as lake of origin.

Mercury levels in walleye and pike from the lakes along the diversion route are generally above the Canadian limit for mercury in fish sold for human consumption in Canada (0.5 ppm). For a number of lakes, the mercury levels in these fish are too high to permit their sale in any market. In contrast to pike and walleye, mercury levels in whitefish from all the diversion lakes are below the Canadian limit.

Only a limited amount of pre-impoundment data exists for mercury levels in fish in the Agreement area; virtually all of these data are for Southern Indian Lake. Analysis of the available data on mercury levels in whitefish from Southern Indian Lake showed a substantial increase immediately after impoundment, followed by noticeable declines over most of the lake from 1978/79 to 1982. Mercury concentrations after 1982 were at or close to concentrations found in 1975 (pre-impoundment).

Analysis of the available data for pike and walleye from Southern Indian Lake suggests that increases in their mercury levels also took place after impoundment. However, unlike the whitefish, this increase has not been followed by a decline. Mercury levels in pike appear to have increased steadily over the post-impoundment period, while walleye appear to have maintained stable mercury levels. No other post-impoundment trends were evident in fish from the other lakes sampled.

Post-impoundment mercury levels in fish from flooded lakes along the diversion route exceeded those of fish from near-by but unflooded lakes. These higher levels did not appear to be related to any higher background mercury levels.

Mercury levels in the piscivorous species (pike and walleye) were higher than in the non-piscivorous fish (whitefish). Within a given species, the older and usually larger fish had higher mercury levels than the younger and smaller fish.

4. Mercury levels in the majority of the people tested in the Agreement area were within the normal range of 0-20 parts per billion (ppb) in blood. However, because of the sensitivity of the fetus to mercury poisoning, there is some concern that a number of women of child-bearing age had mercury levels which exceeded 20 ppb, the generally regarded upper limit of safety.

A voluntary mercury testing program has been offered for the last decade to six native communities located within or near the Agreement area, including South Indian Lake and Nelson House. The majority of people tested in South Indian Lake had mercury levels in the normal range for every year, except 1979 when the majority of the people had levels between 20-99 ppb. The majority of the people tested in Nelson House also had mercury levels in the normal range for every year, except 1978 when the majority of the people had levels between 20-99 ppb.

Comparison of the mercury levels between the six native communities showed that South Indian Lake followed by Nelson House had the highest reported mercury levels. York Landing and Split Lake had intermediate levels significantly below those of Nelson House and South Indian Lake. Cross Lake and Norway House had the lowest reported values, significantly below York Landing and Split Lake.

Comparisons were also made to five communities in Ontario and Quebec which had some of the highest reported values in Canada. The percentage of samples above 99 ppb in these communities were substantially higher than in the six Manitoba communities. However, in terms of the percentage of samples above 20 ppb, two Manitoba communities, South Indian Lake and Nelson House, fell within the range of the five Quebec and Ontario communities.

Of the mercury samples taken from women of child bearing age in Nelson House and South Indian Lake, one value of 150 ppb and one of 125 ppb respectively were recorded. In addition there were a number of samples in the increasing risk range of 20-99 ppb. This is a cause of some concern.

5. Mercury levels in Mink and Otter (which are fish eating mammals) from the Agreement area are elevated in comparison to other Manitoba regions but are below toxic levels.

The mercury concentrations in mink in the Southern Indian Lake area were not significantly different from the Nelson House area, but both these areas were higher than the Split Lake area. No conclusion could be reached regarding long-term trends because of a lack of a continuous time series of data.

6. The elevated mercury levels in the fish from the Diversion lakes resulted from increases in the rate of methyl mercury production. The net production of methyl mercury in the aquatic environment is controlled by bacterial processes.

Methyl mercury is the predominant form of mercury found in fish. Methyl mercury is produced (methylation) and destroyed (demethylation) by a variety of bacteria. The physical, chemical, and biological changes that occurred following the diversion affected the balance of the methylating and demethylating activity in the bacterial community. The net effect of these changes was an increase in methyl mercury.

The net production of methyl mercury is determined by the relative rates of methylation and demethylation (the methylation/demethylation ratio). A distinct seasonal pattern for methylation rates was observed during the sampling period. Evidence of a possible seasonal pattern for demethylation rates was less conclusive. The combined effect on the methylation/demethylation ratio was to produce a strongly peaked seasonal pattern with the highest ratios occuring in late July and August in all the lakes and reservoirs examined.

Methylation rates were generally much higher in areas of flooded terrain than in areas of non-flooded, natural lake sediments. The demethylation rates over flooded terrain were either the same as or lower than those in non-flooded areas. The methylation/demethylation ratios were therefore also higher in flooded than in non-flooded areas, usually being even more elevated than the methylation rates alone.

Strong positive correlations were obtained between methylation rates and mercury concentrations in small fish sampled from a number of lakes in the Agreement area. (Small fish were sampled since they would best reflect the local conditions of each site.) It is also interesting to note that the period of highest methylation rates and methylation/demethylation ratios coincided with the period of the most rapid uptake of mercury into fish.

The importance of two of the major Agreement findings becomes clear when they are restated together: 1. mercury concentrations in fish are strongly correlated with rates of methyl mercury production; and 2. rates of methyl mercury production are highest in areas of flooded terrain. Together these findings explain why the lakes in the Agreement area which were the most extensively flooded contained fish with the highest mercury concentrations.

The methyl mercury that has been produced by the bacterial community is eventually picked up by the fish, either directly from water through the gills and/or from food. Efforts under the Agreement to determine the importance of the transfer of mercury through the food chain were, unfortunately, not successful.

7. The most important factor affecting the net rate of methyl mercury production is the amount and type of flooded material.

Laboratory and field experiments showed that moss, peat, black spruce, prairie sod and freshwater algae all stimulated the production of methyl mercury. The stimulation of methyl mercury production by such a wide variety of organic materials indicates that the effect of flooding on fish mercury levels is a general response and not limited to areas of boreal forest.

It was also found, at least in the case of moss-peat material, that the stimulatory effect is additive. A doubling of the amount of organic material added resulted in a doubling of the amount of mercury which accumulated in the fish. There were differences in the abilities of the tested organic materials to stimulate the bioaccumulation of mercury in fish, e.g. black spruce boughs increased mercury levels in fish to a greater extent than did the moss-peat material.

There was some indication that in the short term moss can inhibit methylation. This inhibition may be of short duration, because as the moss is decomposed and used as a nutrient source by the microbiota, bound mercury can be released and may be methylated.

8. The oxygen level at the methylation site was the most important of a number of other environment factors examined to determine their effect on methylation rates.

Laboratory experiments showed that the net production of methyl mercury was higher when oxygen levels were very low. The most severe oxygen depletion conditions in the Agreement area occurred in the bottom water of Notigi Reservoir during the filling period (1974-76). However conditions have improved and low oxygen concentrations do not now generally occur anywhere in the Agreement area, with the exception of Issett Lake. The low oxygen levels at the bottom of Issett Lake are sufficient to account for the much higher rate of methylation observed here.

Other factors which were examined to

determine their influence on methyl mercury production included sediment, temperature, pH, and the availability of mercury. It is interesting to note that the availability of mercury for methylation did not appear to be a sole controlling factor in the accumulation of mercury in fish muscle.

9. It may take decades for net mercury methylation rates and mercury levels in fish to return to pre-impoundment levels in northern reservoirs.

Methylation rates were measured in an area of Southern Indian Lake from 1982 to 1985 (about ten years after reservoir impoundment). There was no indication of a decline in the methylation rates over flooded terrain in comparison to the rates over the non-flooded, natural lake bottom. The persistence of elevated methylation rates was also demonstrated by a comparison between methylation rates in reservoirs and natural lakes. The ages of the reservoirs varied, with the oldest one being 42 years. The methylation rates measured over areas of flooded terrain in the reservoirs were all high relative to the unflooded, natural lake areas.

The length of time for methylation rates to return to pre-impoundment levels appears to depend on the type of vegetation flooded by reservoir formation. Methylation rates and mercury levels in fish were low in Cedar Lake which was flooded 20 years ago. This lake, located 200 km south of the Agreement area, was in an area of marsh land, and not boreal forests. The much lower loading of nutrients from the flooded material and the apparently higher degradability of this material seem to be responsible for the relative short duration of the mercury problem in this lake.

10. Elevated mercury levels in fish following a reservoir impoundment is a widespread problem. Many factors appear to influence the mercury levels which make it impossible to predict the severity of the problem at a given reservoir, especially if there is no previous experience in the same area.

Two other studies have recently been completed which documented both pre- and post-impoundment levels of mercury in fish, one in Northern Quebec, Canada and one in South Carolina, U.S.A. In both cases, mercury levels in fish rose significantly after impoundment.

Another recently completed study attempted to assess the risk of mercury problems in different reservoirs across Canada. Emphasis was placed on examining factors that differed among the reservoirs, rather than on factors which changed as a result of flooding. No obvious physical or chemical differences among these reservoirs could account for their differences in post-impoundment mercury levels in fish.

RECOMMENDATIONS

1. It is recommended that the results of the Canada-Manitoba Agreement on the study and monitoring of mercury in the Churchill River Diversion should be made widely available.

Particular efforts should be made to inform the residents of the Agreement area, especially those of South Indian Lake and Nelson House. (The ongoing mercury testing program in these communities is one means of disseminating this information.)

2. It is recommended that in addition to the on-going commercial fish monitoring program, periodic monitoring of the fish important to the domestic and sports fishery should be continued until their mercury levels reach acceptable values.

3. It is recommended that fish management activities should be implemented to the maximum extent possible as a means of mitigating the mercury problem.

These fish management activities include recommended consumption of non-predatory and small-sized fish only, compensation for any losses to the commercial fisheries, and the provision of alternative fishing opportunities. Other remedial measures are considered to be too expensive, impractical, and/or unacceptable because of possible negative environmental impacts.

4. It is recommended that elevated mercury levels following impoundment must be considered explicitly in the planning and costing of reservoirs.

Monitoring of pre-impoundment mercury levels in fish in the planned development area must be an essential part of reservoir planning. Additional studies on the factors affecting net methylation rates, in order to better define the estimated period of elevated mercury levels in fish, would assist in determining the ultimate cost of the mercury problem.

GLOSSARY

AEROBIC - living or active in the presence of free oxygen.

ANAEROBIC - living or active in the absence of free oxygen.

ANOXIC - deficiency of oxygen.

ANTHROGENIC - involving the impact of man on nature.

BENTHIC - pertaining to, or relating to **Benthos.**

BENTHOS - **Organisms** that live on or in the bottom of bodies of water.

BIOACCUMULATION the increasing concentration of a substance (eg. mercury) up the food chain.

BIO-MAGNIFICATION - see **Bioaccumulation.**

CHLOROPHYLL - one of a number of green pigments present in plant cells which are essential in the utilization of light energy in photosynthesis.

CORRELATION - the study of the simultaneous variation of two or more variables. If one variable (e.g. mercury concentration) increases as the other increases (eg fish size) and decreases as the other decreases, there is said to be a positive correlation.

DEMETHYLATION - conversion of one type of chemical compound into another by the removal of the methyl group (CH^{3}-). The reverse of methylation.

GLACIO-LACUSTRINE - of, relating to, or coming from lakes deriving much or all of their water from the melting of a glacier.

INORGANIC - not **Organic**, not containing carbon or compounds of carbon (with a few exceptions).

ISOTOPE - one of a set of chemically identical types of atoms. A radioactive isotope of mercury was used to provide a method for following the movement of mercury through the food chain.

LIMNOCORRALS - large enclosures to isolate a defined water mass from a larger body of water. The limnocorrals used were approximately 10 meters in diameter and 3-4 meters deep.

MERCURIC CHLORIDE - an **Inorgainic** form of mercury.

MERCURY — a naturally occuring chemical element. It can exist in three different forms; as the pure element, a silver-white liquid often used in thermometers; in an Inorganic form, such as mercuric sulfide (HgS) found in bedrock; and in an ORGANIC form, such as methyl mercury (CH^{3}Hg-) found in fish.

METHYLATION - conversion of one type of chemical compound into another by the addition or incorporation of the methyl group (CH^{3}Hg)+) e.g. conversion of **Inorganic** mercury to **Methyl-Mercury**.

METHYL MERCURY -an **Organic** form of mercury, containing the methyl group (CH^{3}-). It has a high affinity for animal tissue and is concentrated up the food chain. The correct name is methyl mercury ion (CH^{3}HG)+, but for ease of reference in this report, methyl mercury is used.

MICROBES - very minute **Organism.**

MICROBIAL - pertaining to Microbes.

NORTHERN FLOOD AGREEMENT - a 4 party agreement signed in 1977 to ensure that all persons adversely affected by the Lake Winnipeg, Churchill-Nelson Hydro-electric Development Project receive fair and equitable treatment.

NUTRIENT - any substance which promotes growth or provides energy for a living **Organism.**

ORGANIC - relating to or derived from living **Organisms.**

ORGANISM - a living being.

PERMAFROST - ground that is permanently frozen.

pH - a measure of the acidity or alkalinity of a liquid. pH values can range from 0 (highly acidic) to 14 (highly alkaline) with a pH reading of 7 indicating a neutral solution.

PISCIVOROUS - feeding on fish.

PLANKTON - the passively floating or weakly swimming animal and plant life of a body of water consisting chiefly of minute plants and of minute animals that have only weak powers of movement.

RADIOACTIVE - radioactivity is the property exhibited by unstable **Isotopes** of elements which decay emitting radiation.

REGRESSION ANALYSIS - a relationship between 2 or more correlated variables that is often determined from data and is used especially to predict values of one variable when given the values of the others.

SPECIFIC ACTIVITY - for a sample of radioactive element or compound, specific activity is a measure of the proportion of the element which is in a **Radioactive** form **(Isotope)**. The units are radioactivity per weight of the element in the sample.

SPECIFIC RATE - the percent of available mercuric ion methylated or methyl mercury ion demethylated.

SUBSTRATE - **Nutrient** medium for bacteria.

VOLCANOGENIC - produced by a volcano.

ZOOPLANKTON - animal life of the **Plankton**.

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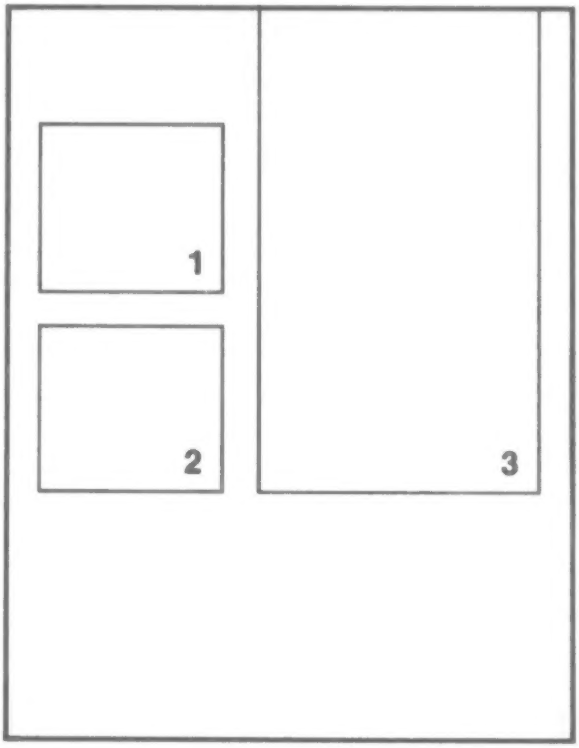

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Canada - Manitoba Agreement on the Study and Monitoring of Mercury in the Churchill River Diversion

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Volume 1

MERCURY IN WATER AND SEDIMENTS IN THE CHURCHILL AND NELSON RIVERS MANITOBA, CANADA

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Water Standards and Studies Report #86-3

SPECIAL NOTE

This report was prepared for the Canada-Manitoba Agreement on the Study and Monitoring of Mercury in the Churchill River Diversion. Any conclusions, recommendations, or opinions expressed herein are those of the author and are not necessarily the same as those of the Steering Committee for this Agreement.

SUMMARY

Concentrations of extractable mercury in water and total mercury in river and lake surficial bottom sediments were measured in samples collected from areas affected by stream regulation projects in the Churchill and Nelson Rivers. The data base extends from approximately 1977 to 1984 for mercury in water and from 1979 to 1983 for mercury in bottom sediments. This data was generated as part of a routine water quality monitoring activity within the Province of Manitoba.

Most extractable mercury in water observations were below the analytical limit of quantitation (0.02 µg/L). The few exceptions were likely either influenced by sampling or analytical bias or were representative of the upper percentiles of the normal distribution of extractable mercury in water within this area.

Total mercury concentrations in surficial bottom sediments displayed little apparent change at all sites throughout the duration of the investigation. Similarly, little difference was apparent between the various sites examined within areas affected by the Churchill River diversion, within areas affected by Lake Winnipeg regulation and at one unaffected site.

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Introduction

Data for a variety of physical and chemical water quality variables within several aquatic media were collected from a wide geographical area within Manitoba. The information generated from this routine water quality monitoring activity supported environmental management programs. Data uses are numerous and include for example, delineation of long-term changes and trends in water quality, characterization of background or ambient water quality or relating observed water quality trends to pollution abatement programs or to environmental interventions.

It was thought that a portion of the data generated by this monitoring activity would add to the information base regarding the presence and distribution of mercury within areas affected by the Churchill River diversion. The collection and assessment of this information, identified as Project 1.1, was not cost-shared under the Canada-Manitoba Agreement on the Study and Monitoring of Mercury in the Churchill River Diversion, since it was part of an ongoing program of the Manitoba Department of Environment and Workplace Safety and Health.

Water quality monitoring activities within northern Manitoba commenced in the early 1970's with joint Canada-Manitoba studies being undertaken under the guidance of the Lake Winnipeg, Churchill and Nelson Rivers Study Board. Subsequent to these early studies, program evolution occurred such that monitoring continued at some original sites, monitoring was discontinued at other sites and monitoring was initiated at further locations. Parametric coverage also changed in accordance with perceived needs and changes in analytical technologies.

While studies under this mercury agreement were specific to areas affected by the Churchill River diversion projects, the information discussed within this report also include data collected from the upper Nelson River. Hence, the entire general geographical area encompassed by the Northern Flood Agreement was included.

Mercury in natural waters can exist as elemental mercury (0), the mercurous form (+1) or as the mercuric state (+2). Mercury speciation is dependent upon pH, redox potential and the characteristics of the anions available for stable bonding (Moore and Ramamoorthy, 1983). The occurrence of mercury in water can result from both natural and anthropogenic sources. Natural sources include mercury-bearing sulfide minerals such as cinnabar, metacinnabar, and sphalerite as well as elemental mercury. Anthropogenic sources include industrial and municipal wastewater including wastes produced from the smelting of ores to extract metals such as lead, zinc and copper (Faust and Aly, 1980).

Both aerobic and anerobic bacteria can methylate divalent inorganic mercury into methyl mercury (National Research Council, 1978). Methylated mercury is more soluble in water and is more biologically active in water than either elemental mercury or divalent inorganic mercury. Consequently, it is the methylated mercury forms that are of most concern within aquatic systems because of their propensity to biomagnify throughout the food chain. Once mercury is incorporated within biological tissues, attenuation is slow.

The measurement of extractable mercury in water within this routine monitoring activity was designed to detect material increases in the combined inorganic and organic forms of mercury present, should such increases be of sufficient magnitude to cause the routine positive detection of mercury in water. Unfortunately, an alteration of the ratio of inorganic to organic forms of mercury within these aquatic systems, caused by changes in the rate of methylation-demethylation, would not be detected.

Moore and Ramamoorthy (1983) summarized the major environmental processes affecting mercury within suspended and bed sediments. Both sorption and bonding correlated best with the organic content of sediments. Mercury can be associated with sediments to various degrees ranging for example, from weak Van der Waals forces to strong covalent bonding, co-precipitation with iron-manganese nodules and incorporation in crystal lattices. Mercury can also be released from sediments into the interstitial and surficial waters, dependent upon both physical and chemical properties of the sediment and water. Munawar et al. (1984) reported that sediments act as a reservoir for trace metals in aquatic environments. This has been substantiated by numerous investigations. For example, 97% of the mercury within the Ottawa River was associated with bed sediments (Kudo et al. 1977).

Moore and Ramamoorthy (1983) further reported that partitioning between mercury in suspended sediments and mercury in water have been calculated to be approximately 1×10^5 times. Thus, given knowledge of the mercury content of suspended sediments, it is possible to estimate the concentration of dissolved mercury.

Knowledge of the temporal changes of mercury concentrations within bed sediments is a useful tool for monitoring both the redistribution of mercury within riverine systems and the possible material increase of mercury into any aquatic system from anthropogenic sources such as base-metal mining and smelting operations. However, this information alone may not provide the necessary information for water quality managers to anticipate mercury associated problems with biomagnification through the aquatic food chain. The biological transformation of mercury within bed sediments can occur at rates which are independent of either the redistribution of mercury or the anthropogenic influenced increase of sediment related total mercury. The principal previously mentioned biological-related transformation, namely methylation and demethylation, can readily alter the sediment associated inorganic mercury to the more readily biomagnified organic mercury without substantially altering the total mercury concentration.

Methodology

Analyses for mercury in water at most sites affected by the Churchill River diversion were initiated during the summer of 1977 or during the winter of 1978 and continued until March, 1984. However, sampling was somewhat variable. For example, insecure ice conditions near the community of South Indian Lake precluded the safe collection of samples from ski-equipped aircraft during winter months. Therefore, an alternate location within the South Basin of Southern Indian Lake was chosen. An alternate sampling site at Apussigamasi Lake on the Burntwood River was also used for the same reason.

It should be noted that, while the Barrington River upstream of Opachuanau Lake is geographically located upstream of the point of diversion, it is nevertheless influenced by water level fluctuations resulting from the operation of the diversion. Further, Footprint Lake is not located directly on the diversion route but is influenced through tributary streams.

Water samples for mercury analyses were collected in triplicate in 100 mL Soveril pyrex glass containers from approximately 0.5 m below the water surface. The Soveril pyrex glass sample containers had previously been subjected to heat at 105°C for 4 to 5 hours in order to ensure that all trace of mercury was removed by vapourization. The samples were immediately preserved with the addition of 1 mL of concentrated nitric acid and 1 mL of 5% potassium dichromate. The samples were cooled with ice, then shipped to the W. M. Ward Technical Services Laboratory in Winnipeg for analyses. Samples were received within twenty-four hours of collection; analyses were completed within fourteen days of collection.

The principle of the analytical method has been described by Sorba et al. 1980: The sample containers were allowed to stand undisturbed for several hours, then an aliquot was decanted for analyses. The organic mercury compounds in the aliquot were oxidized to inorganic mercury by heating with sulphuric acid, potassium permanganate and potassium persulfate at a high temperature within a closed system. Stannous chloride in a hydroxylamine sulphate-sodium chloride solution was added in order to reduce all mercuric compounds to elemental mercury. The elemental mercury was passed through a 30 cm Pharmacia UV photometer. Absorbance was measured at 253.7 nm. Mercury was reported as extractable. The analytical limit of quantitation was 0.02 µg/L, with the exception of August and September, 1977, when the limit was 0.05 µg/L.

Surficial bottom sediment samples were obtained with the use of a standard Wildco Ponar dredge. Sediment sampling was initiated in 1979 and continued until 1983. Samples were collected in triplicate, where possible in 1979, 1980, 1981 and 1983. Single samples only were collected in 1982. The top 1 to 3 cm of sediment were removed, placed in whirl-pac bags, and shipped to the W. M. Ward Technical Services Laboratory in Winnipeg for analyses. Samples were stored frozen until analyses could be completed.

The analysis was carried out in accordance with the method described by Sorba et al. (1980). Inorganic forms of mercury were extracted and organic forms were oxidized in a 1 g (wet weight) sediment aliquot by digestion with sulphuric acid, nitric acid, hydrochloric acid, potassium permanganate and potassium persulphate. The clear supernatant was then analyzed using a similar procedure as described previously for determining extractable mercury concentrations in water. The analytical limit of quantitation was 0.02 µg/g (wet weight). The results were reported as dry weight equivalent total mercury.

Results and Discussion

Mercury in Water

The sampling locations are shown in Figure 1, and are described in Table 1. Extractable mercury data in water are presented in Table 2.

With few exceptions all extractable mercury concentrations in water samples collected from areas affected by the Churchill River diversion, and from one upstream unaffected site were less than the limit of quantitation (0.02 µg/L). The occasional concentrations

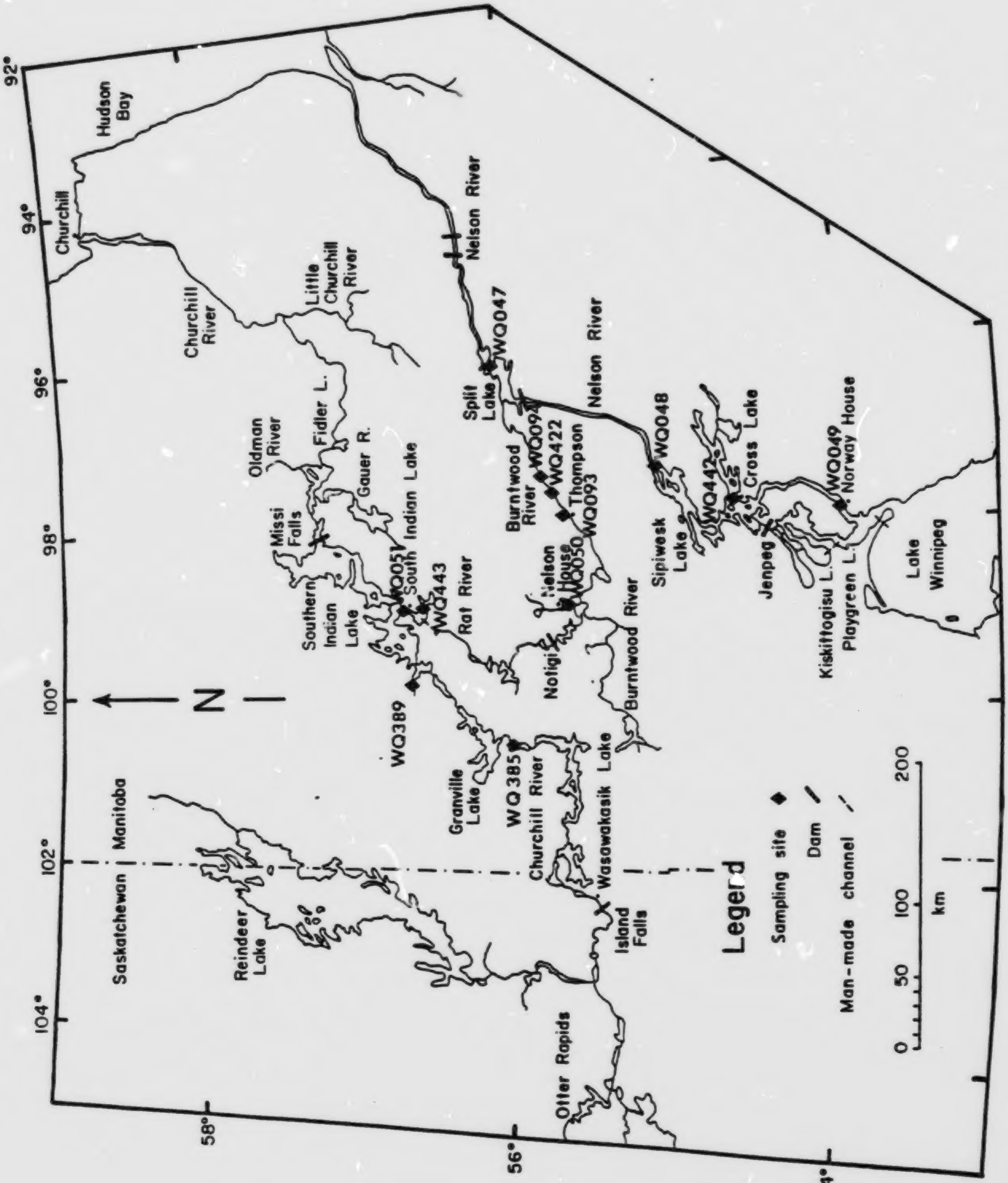
58°
56°
54°
104°
102°
100°
98°
96°
94°
92°
N
Saskatchewan
Manitoba
Reindeer Lake
Hudson Bay
Churchill
Churchill River
Little Churchill River
Nelson River
Oldman River
Fidler L.
Gauer R.
Missi Falls
South Indian Lake
Southern Indian Lake
Rat River
Split Lake
Burntwood River
Thompson
Nelson House
Notigi
Burntwood River
Sipiwesk Lake
Cross Lake
Norway House
Jenpeg
Kiskittogisu L.
Playgreen L.
Lake Winnipeg
Granville Lake
Churchill River
Wasawakasik Lake
Island Falls
Otter Rapids
WQ047
WQ094
WQ422
WQ093
WQ050
WQ048
WQ442
WQ049
WQ051
WQ443
WQ389
WQ385
Legend
Sampling site ◆
Dam
Man-made channel
0
50
100
200
km

Figure 1: Location of sampling sites.

Table 1: Description of sampling locations.

	Upstream - Churchill River Diversion
WQ 385	Churchill River upstream of Granville Lake. Lat. 56:08; Long. 100:30
	Influenced - Churchill River Diversion
WQ 389	Barrington River upstream of Opachuanau Lake Lat. 56:45; Long. 99:58
WQ 051	Southern Indian Lake near the community of South Indian Lake. Lat. 56:46; Long. 98:57.
WQ 443	Southern Indian Lake near the centre of the South Bay. Lat. 56:44; Long. 99:00
WQ 050	Footprint Lake near the community of Nelson House. Lat. 55:46; Long. 98:54
WQ 093	Burntwood River at Thompson Lat. 55:45; Long. 97:50
WQ 422	Burntwood River at Apussigamasi Lake, inlet area. Lat. 55:49; Long. 97:42
WQ 094	Burntwood River at Apussigamasi Lake, central area. Lat. 55:51; Long. 97:36
WQ 047	Split Lake near the community of Split Lake. Lat. 56:14; Long. 96:05
	Upper Nelson River
WQ 049	Little Playgreen Lake near the community of Norway House. Lat. 53:59; Long. 97:50
WQ 442	Cross Lake near the community of Cross Lake. Lat. 54:37; Long. 97:47
WQ 048	Sipiwesk Lake, near outlet Lat. 55:13; Long. 97:20

Table 2: Extractable mercury concentrations in surface waters within the study area.

WQ 385 Churchill River up-stream of Granville Lake		WQ 389 Barrington River up-stream of Opachuanau Lake		WQ 051 Southern Indian Lake near the Community of South Indian Lake		WQ 443 Southern Indian Lake near the Centre of South Bay		WQ 050 Footprint Lake Near the Community of Nelson House		WQ 093 Burntwood River at Thompson	
Date	µg(Hg/L	Date	µg(Hg)/L	Date	µg(Hg)/L	Date	µg(Hg)/L	Date	µg(Hg)/L	Date	µg(Hg)/L
78 02 09	0.02 L*	78 02 09	0.02 L	77 06 07	0.02	78 02 07	0.02 L	77 06 07	0.03 L	77 08 16	0.05 L
78 03 09	0.02 L	78 03 09	0.02 L	77 07 18	0.02 L	78 03 09	0.02 L	77 07 18	0.02 L	77 09 13	0.05 L
78 06 07	0.02 L	78 06 07	0.02 L	77 08 16	0.05 L	79 01 08	0.02 L	77 09 13	0.05 L	77 10 19	0.04
78 07 10	0.02 L	78 07 10	0.02 L	77 09 13	0.05 L	79 02 05	0.02 L	78 02 09	0.02 L	78 02 08	0.02 L
78 08 17	0.02 L	78 08 17	0.02 L	78 06 07	0.02 L	79 03 05	0.02 L	78 03 09	0.02 L	78 03 08	0.02 L
78 09 11	0.02	78 09 11	0.02	78 07 10	0.02 L	80 01 07	0.02 L	78 06 07	0.02 L	78 06 06	0.02 L
78 10 18	0.02 L	78 10 18	0.02 L	78 08 17	0.02 L	80 02 11	0.02 L	78 07 10	0.02 L	78 07 10	0.02 L
79 01 08	0.02 L	79 01 08	0.02 L	78 09 11	0.02	80 03 11	0.02 L	78 08 16	0.02 L	78 08 16	0.02 L
79 02 05	0.02 L	79 02 05	0.02 L	78 10 18	0.02 L	81 01 20	0.02 L	78 09 11	0.02	78 09 12	0.02
79 03 05	0.02 L	79 03 05	0.02 L	79 06 12	0.02 L	81 03 18	0.02 L	78 10 18	0.02 L	78 10 17	0.02 L
79 06 12	0.02 L	79 06 12	0.02 L	79 07 09	0.02 L	82 01 13	0.02 L	79 01 08	0.02 L	79 01 10	0.02 L
79 08 13	0.02 L	79 07 09	0.02 L	79 08 13	0.02 L	82 02 10	0.02 L	79 02 05	0.02 L	79 02 07	0.02 L
79 09 10	0.02 L	79 08 13	0.02 L	79 09 10	0.02 L	84 03 07	0.02 L	79 03 05	0.02 L	79 03 05	0.02 L
79 10 16	0.02 L	79 09 10	0.02 L	79 10 15	0.02 L			79 06 12	0.02 L	79 06 12	0.02 L
80 01 08	0.02 L	79 10 16	0.02 L	80 06 09	0.02 L			79 07 10	0.02 L	79 07 10	0.02
80 02 12	0.02 L	80 01 08	0.02 L	80 07 07	0.02 L			79 08 13	0.02 L	79 08 15	0.02 L
80 03 11	0.02 L	80 02 12	0.02 L	80 08 11	0.02 L			79 09 10	0.02 L	79 09 10	0.02 L
80 06 09	0.02 L	80 03 11	0.02 L	80 09 09	0.02 L			79 10 15	0.02 L	79 10 17	0.02 L
80 07 07	0.02 L	80 06 09	0.02 L	81 06 01	0.02 L			80 01 08	0.02 L	80 01 09	0.02 L
80 08 11	0.02	80 07 07	0.02 L	81 07 27	0.02 L			80 02 12	0.02 L	80 02 11	0.02 L
80 09 09	0.02 L	80 08 11	0.02 L	81 10 07	0.02 L			80 03 10	0.02 L	80 03 13	0.02 L
81 01 20	0.02 L	80 09 09	0.02	82 06 10	0.02 L			80 06 09	0.02 L	80 06 09	0.02 L
81 06 01	0.02 L	81 01 20	0.02 L	82 07 06	0.02 L			80 07 07	0.02 L	80 07 07	0.02 L
81 07 27	0.02 L	81 03 18	0.02 L	82 08 18	0.02 L			80 08 11	0.02 L	80 08 11	0.02 L
81 10 07	0.02 L	81 06 01	0.02 L	83 07 06	0.02 L			80 09 09	0.02	80 09 11	0.02 L
82 01 13	0.02 L	81 07 27	0.02 L	83 08 24	0.02 L			81 01 20	0.02 L	81 01 20	0.02 L
82 02 10	0.02 L	81 10 07	0.02 L	84 02 06	0.02 L			81 02 19	0.02 L	81 02 19	0.02 L
82 06 10	0.02 L	82 01 13	0.02 L					81 03 18	0.02 L	81 03 17	0.02 L
82 07 06	0.02 L	82 02 10	0.02 L					81 06 01	0.02 L	81 06 01	0.02 L
82 08 18	0.02 L	82 06 10	0.02 L					81 07 27	0.02 L	81 07 27	0.02 L
83 07 06	0.02 L	82 07 06	0.02 L					81 10 07	0.02 L	81 10 06	0.02 L
83 08 24	0.02 L	82 08 18	0.02 L					82 01 13	0.02 L	82 01 12	0.02 L
83 09 20	0.02 L	83 07 06	0.02 L					82 02 10	0.02 L	82 03 10	0.02 L
84 01 09	0.02 L	83 08 24	0.02 L					82 06 10	0.02 L	82 06 09	0.02 L
84 03 07	0.02 L	83 09 20	0.02 L					82 07 06	0.02 L	82 07 08	0.02 L
		84 02 06	0.02 L					82 08 18	0.02 L	82 07 08	0.02 L
		84 03 07	0.02 L					83 07 06	0.02 L	82 08 17	0.02 L
								83 08 24	0.02 L	83 07 05	0.02 L
								83 09 20	0.02 L	83 08 23	0.02 L
								84 01 09	0.02 L	83 09 19	0.02 L
								84 02 06	0.02 L	84 01 11	0.02 L
								84 03 07	0.02 L	84 02 06	0.02 L
										84 03 06	0.02 L

* Values qualified by a "L" were less than the technical limit of quantitation.

Table 2: continued

WQ 422 Burntwood River at Apussigamasi Lake, Inlet Area		**WQ 094** Burntwood River at Apussigamasi Lake, Central Area		**WQ 047** Split Lake near the Community of Split Lake		**WQ 049** Little Playgreen Lake near the Community of Norway House		**WQ 442** Cross Lake near the Community of Cross Lake		**WQ 048** Sipiwesk Lake near Outlet Area	
Date	µg(Hg)/L	Date	µg(Hg)/L	Date	µg(Hg)/L	Date	µg(Hg)/L	Date	µg(Hg)/L	Date	µg(Hg)/L
79 06 12	0.02 L			77 06 08	0.02 L	77 06 06	0.02 L	78 02 06	0.02 L	77 06 06	0.02 L
79 07 10	0.02 L	77 08 15	0.05 L	77 07 18	0.02 L	77 07 19	0.03	78 03 07	0.02 L	77 07 19	0.03
79 08 13	0.02 L	77 09 13	0.05 L	77 08 15	0.05 L	77 08 17	0.05 L	78 06 05	0.02 L	77 08 17	0.05 L
79 09 10	0.02 L	77 10 19	0.02 L	77 09 13	0.05 L	77 09 12	0.05 L	78 07 13	0.02 L	77 09 12	0.05 L
79 10 17	0.02 L	78 02 06	0.02 L	78 02 08	0.02 L	78 02 06	0.02 L	78 08 15	0.03	78 02 06	0.02 L
80 06 09	0.02 L	78 03 09	0.02 L	78 03 09	0.02 L	78 03 07	0.02 L	78 09 12	0.02 L	78 03 07	0.02 L
80 07 07	0.02 L	78 06 06	0.02 L	78 06 06	0.02 L	78 06 05	0.02 L	78 10 16	0.02 L	78 06 05	0.02 L
80 08 11	0.02 L	78 07 10	0.02 L	78 07 10	0.02	78 07 13	0.02 L	79 01 09	0.02 L	78 07 13	0.02 L
80 09 09	0.02 L	78 08 16	0.02 L	78 08 17	0.02 L	78 08 15	0.03	79 02 07	0.02 L	78 08 15	0.02 L
81 06 04	0.02 L	78 09 12	0.02 L	78 09 11	0.02	78 09 12	0.02	79 06 13	0.02 L	78 09 12	0.02
81 07 28	0.02 L	78 10 18	0.02 L	79 01 10	0.02 L	78 10 16	0.02 L	79 07 10	0.02 L	78 10 16	0.02 L
81 10 07	0.02 L	79 01 10	0.02 L	79 02 07	0.02 L	79 01 09	0.02 L	79 08 14	0.02 L	79 01 09	0.02 L
82 06 09	0.02 L	79 02 07	0.02 L	79 03 05	0.02 L	79 02 07	0.02 L	79 10 17	0.02 L	79 02 07	0.02 L
82 07 06	0.02 L	79 03 05	0.02 L	79 06 12	0.02 L	79 03 06	0.02 L	80 01 09	0.02 L	79 03 06	0.02 L
82 08 17	0.02 L	80 01 09	0.02 L	79 08 13	0.02 L	79 06 13	0.02 L	80 02 14	0.02 L	79 06 13	0.02 L
83 07 05	0.02 L	80 02 11	0.02 L	79 09 10	0.02 L	79 07 10	0.02 L	80 03 13	0.02 L	79 07 10	0.02 L
83 08 23	0.02 L	80 03 10	0.02 L	79 10 15	0.02 L	79 08 14	0.02 L	80 06 10	0.02 L	79 08 14	0.02 L
83 09 19	0.02 L	81 01 20	0.02 L	80 01 07	0.02 L	79 10 17	0.02 L	80 07 08	0.02 L	79 10 15	0.02 L
		81 03 18	0.02 L	80 02 11	0.02 L	80 01 09	0.02 L	80 08 13	0.02 L	80 01 09	0.02 L
				80 03 10	0.02 L	80 02 14	0.02 L	80 09 10	0.02 L	80 02 14	0.02 L
				80 06 09	0.02 L	80 03 13	0.04	81 01 22	0.02 L	80 03 13	0.03
				80 07 07	0.02 L	80 06 10	0.02 L	81 02 16	0.02 L	80 06 10	0.02 L
				80 08 11	0.02 L	80 07 08	0.02 L	81 03 16	0.02 L	80 07 08	0.02 L
				80 09 09	0.02 L	80 08 13	0.02	81 06 02	0.02 L	80 08 13	0.02 L
				81 01 20	0.02 L	81 09 10	0.02 L	81 07 30	0.02 L	80 09 10	0.02 L
				81 03 18	0.02 L	81 01 22	0.02 L	81 10 05	0.02 L	81 01 22	0.02 L
				81 06 04	0.02 L	81 02 16	0.02 L	82 01 11	0.02 L	81 02 16	0.02 L
				81 07 28	0.02 L	81 03 16	0.02 L	82 03 08	0.02 L	81 03 16	0.02 L
				81 10 08	0.02 L	81 06 02	0.02 L	82 06 08	0.02 L	81 06 02	0.02 L
				82 01 13	0.02 L	81 07 30	0.02 L	82 07 07	0.02 L	81 07 30	0.02 L
				82 02 10	0.07	81 10 05	0.02 L	82 08 16	0.02 L	81 10 05	0.02 L
				82 06 09	0.02 L	82 02 08	0.02 L	83 07 04	0.02 L	82 01 11	0.02 L
				82 07 06	0.02 L	82 03 08	0.04	83 08 22	0.02 L	82 02 08	0.11
				82 08 17	0.02 L	82 06 08	0.02 L	83 09 22	0.02 L	82 03 08	0.02 L
				83 07 05	0.02 L	82 07 07	0.02 L	84 01 10	0.02 L	82 06 08	0.02 L
				83 08 23	0.02 L	82 08 16	0.02 L	84 02 08	0.02 L	82 07 07	0.02 L
				83 09 19	0.02 L	83 07 04	0.02 L	84 03 05	0.02 L	82 08 16	0.02 L
				84 02 09	0.02 L	83 08 22	0.02 L			83 07 04	0.02 L
				84 03 06	0.02 L	83 09 22	0.02 L			83 08 22	0.02 L
						84 01 10	0.02 L			83 09 22	0.02 L
						84 02 08	0.02 L			84 01 10	0.02 L
						84 03 05	0.02 L			84 07 08	0.02 L
										84 03 05	0.02 L

detected were reported as 0.02 µg/L, with 0.07 µg/L being observed on a single occasion.

Because most of the detectable concentrations were observed on a single sampling excursion, it would appear likely that such values resulted from either a systematic sampling or analytical bias. Other extractable mercury values may have similarly been influenced. Experience with the collection and analyses of water samples for extractable mercury content has shown that, due to the persistent and ubiquitous nature of mercury in uncontrolled field surroundings, sample contamination may sometimes occur. Conversely however, extractable mercury has been detected in water at concentrations near 0.02 µg/L downstream of anthropogenic activities such as metal mining and smelting operations. Hence, it is possible that the observed positive detection of extractable mercury concentrations in water were real and represented the upper percentiles of the normal distribution of mercury.

Water samples, collected from three sites located on the upper Nelson River namely, Little Playgreen Lake near Norway House, Cross Lake near the community of Cross Lake and near the outlet of Sipiwesk Lake, were analyzed for extractable mercury.

Most observations were below the limit of quantitation (0.02 µg/L). However, concentrations at or above this limit, ranging from 0.02 µg/L to 0.11 µg/L, were reported from several locations at infrequent sampling intervals. The highest concentration detected, namely 0.11 µg/L, was observed at Sipiwesk Lake in February, 1982. Mercury was also detected in samples collected two days later from Split Lake. Such detected extractable mercury concentrations likely occurred for reasons similar to those discussed previously with regard to detected extractable mercury concentrations within areas affected by the Churchill River diversion.

Mercury in Sediments

Total mercury data in lake and river surficial bottom sediments are presented in Table 3. The predominant physical characteristics of the sediment samples were observed during sample collection and are summarized in Table 4. Mercury in surficial bottom sediment monitoring sites were co-located with mercury in water sampling sites (Figure 1; Table 1).

Mercury concentrations in surficial bottom sediments collected from areas affected by impoundment and diversion projects on the Churchill River system were variable. All concentrations were found to be within the range of "naturally-occurring" mercury in bed sediments as summarized by Williamson (1980).

Mercury concentrations in surficial bottom sediments were regressed with time in order to determine whether the observed changes in such concentrations throughout the duration of the investigation were linearly related with time. This procedure however, was confounded by the large number of data qualified with less than quantitation designations. Two methods were used to assist the interpretation of these data. Firstly, the numerical values recorded in Table 3 were entered into the regression analyses without consideration of the qualifying designations. The qualifying designations were then used to assist in subjectively assessing the validity of the regression analyses. This method resulted in the

Table 3: Total mercury content of river and lake surficial bottom sediments.
Results are expressed as µg/g, dry weight equivalent.

	LOCATION	1979	1980	1981	1982	1983
WQ 385	Churchill River upstream of Granville Lake	<0.02*	0.02	<0.04	<0.03	<0.03
		0.02	0.03	0.03	-	<0.04
		0.02	0.03	<0.03	-	<0.04
WQ 389	Barrington River upstream of Opachuanau Lake	0.07	0.06	0.07	<0.05	<0.06
		0.06	0.07	0.07	-	<0.05
		0.07	0.09	<0.04	-	0.05
WQ 051	Southern Indian Lake near South Indian Lake Community	0.05	0.04	<0.03	<0.04	<0.04
		0.05	0.05	<0.04	-	<0.04
		0.04	0.06	<0.04	-	<0.04
WQ 050	Footprint Lake near Nelson House Community	0.08	0.07	0.06	<0.03	<0.04
		0.08	0.09	0.06	-	<0.04
		0.08	0.07	<0.04	-	<0.03
WQ 093	Burntwood River at Thompson	0.02	-	<0.03	<0.03	<0.03
		-	-	<0.03	-	<0.03
		-	-	<0.03	-	<0.03
WQ 422	Burntwood River at Apussigamasi Lake	0.04	0.04	<0.03	<0.04	<0.03
		0.04	0.03	<0.03	-	<0.03
		0.04	0.04	<0.03	-	<0.03
WQ 047	Split Lake near Split Lake Community	0.03	0.04	<0.03	<0.04	<0.04
		0.04	0.04	<0.03	-	<0.04
		0.04	0.05	<0.03	-	<0.04
WQ 049	Nelson River at Norway House Community	0.03	0.03	0.03	0.06	<0.06
		0.04	0.04	0.05	-	<0.07
		0.03	0.04	0.05	-	<0.05
WQ 442	Cross Lake near Cross Lake Settlement	0.04	0.05	<0.03	0.06	<0.09
		0.04	0.04	0.06	-	<0.04
		0.04	0.04	0.03	-	<0.04
WQ 048	Sipiwesk Lake near outlet	-	0.02	<0.03	<0.03	<0.04
		-	0.03	<0.03	-	<0.04
		-	0.03	<0.03	-	<0.04

* Please note: While the wet weight analytical limit of quantitation was 0.02 µg/g, conversion to dry weight equivalent basis by consideration of individual sample moisture content resulted in the various reported limits of quantitation.

Table 4: Physical characteristics of river and lake surficial bottom sediment samples.

	Location	Characteristics
WQ 385	Churchill River upstream of Granville Lake	Fine mineral sediment; some sand and gravel present; evidence of recent deposition (stratification).
WQ 389	Barrington River upstream of Opachuanau	Fine mineral-organic sediment; evidence of recent deposition (stratification).
WQ 051	Southern Indian Lake near South Indian Lake Community	Fine mineral-sediment; evidence of recent deposition (1 cm of fine clay).
WQ 050	Footprint Lake near Nelson House Community	Fine mineral sediment; evidence of recent deposition (stratification).
WQ 093	Burntwood River at Thompson	Fine mineral sediment; presence of sand; some organic material present.
WQ 422	Burntwood River at Apussigamasi Lake	Fine mineral sediment; evidence of recent deposition (stratification).
WQ 047	Split Lake near Split Lake Community	Fine mineral sediment; evidence of recent deposition (stratification).
WQ 049	Nelson River at Norway House Community	Fine mineral sediment; evidence of recent deposition (stratification).
WQ 442	Cross Lake near Cross Lake Settlement	Fine mineral-organic sediment; evidence of recent deposition (stratification).
WQ 048	Sipiwesk Lake near Outlet	Fine mineral sediment; evidence of recent deposition (stratification).

identification of a statistically significant decreasing linear trend with time at Footprint Lake near the community of Nelson House ($r^2 = 0.86$, slope = $- 0.013$, $\rho \neq 0$, $\alpha = 0.05$). This procedure however, may have artificially reduced the variability of the data, thus resulting in a spurious trend (Type I statistical error).

Secondly, random values less than the qualified concentrations were substituted into the statistical regression analysis in place of the qualified concentrations. The decreasing linear trend with time at Footprint Lake near the community of Nelson House was no longer statistically significant ($\alpha = 0.05$) with this method. This latter method however, may have conversely artificially increased the variability of the data, resulting in the absence of a trend when one may have existed (Type II statistical error).

Analyses of variance and Duncan's Multiple Range tests were used to determine whether differences in total mercury content of surficial bottom sediments existed between sampling locations. The data qualified by less than quantitation designations were treated in a similar manner as previously described with regard to the regression analyses. Numerous statistically significant differences between sites were identified when the qualifying designations were omitted. However, when the analyses were repeated with substituted random variables less than the limit of quantitation, statistical significance was lost.

In summary, uncertainties with rigorous data analysis precludes the identification of differences in total mercury concentrations in surficial bottom sediments with time or between sampling sites with confidence. In any case, the range of total mercury concentrations in surficial bottom sediments within replicates on each sampling occasion, between sampling occasions at each sampling site and between sampling sites included in the investigation was found to be relatively small (<0.02 µg/g to 0.09 µg/g) and near the technical limit of quantitation.

Conclusions

It was concluded that:

(1) Extractable mercury concentrations in water, with limited exceptions, were below the analytical limit of quantitation.

(2) The limited number of extractable mercury concentrations in water observed above the analytical limit of quantitation may have resulted either from systematic or non-systematic analytical or sampling bias or may represent the upper several percentiles of the normal extractable mercury distribution.

(3) Total mercury concentrations in surficial bottom sediments displayed little apparent change at all sampling sites throughout the duration of the investigation.

(4) Little apparent difference was evident between total mercury concentrations in surficial bottom sediments measured at the various sampling sites.

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SURVEY OF EXISTING DATA ON LEVELS AND SOURCES OF MERCURY WITHIN THE REGION COVERED BY THE CANADA-MANITOBA MERCURY AGREEMENT

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1984, December 14

SPECIAL NOTE

This report was prepared for the Canada-Manitoba Agreement on the Study and Monitoring of Mercury in the Churchill River Diversion. Any conclusions, recommendations, or opinions expressed herein are those of the authors and are not necessarily the same as those of the Steering Committee for this Agreement.

EXECUTIVE SUMMARY

1. Lakes with average fish mercury concentrations in excess of commercial standards are common within north-central Manitoba. In about 40% of the lakes for which data are available, average mercury concentrations in northern pike exceed the standard for domestic sale and about 8% exceed the export standard. In contrast, data from northern Saskatchewan indicate that only 9% of the surveyed lakes exceed the domestic standard and the median lake average concentration in northern pike is only about half that for northern Manitoba.

2. The highest concentrations occur in water bodies along the Churchill River Diversion-Burntwood River system and in the region south and southwest of Lynn Lake. Below-average concentrations occur in fish in lakes near Flin Flon and east of the Nelson River.

3. There is no evidence to indicate that significant quantities of mercury are transported into the region, either by human or natural agencies and it is concluded that the elevated concentrations observed in fish are caused by factors which are internal to the region.

4. The bedrock in the region contains mercury in concentrations which are elevated with respect to most other regions in Canada of similar lithology. Particularly elevated values occur in association with sulphide minerals, and volcanogenic host rocks for sulphide mineralization are widespread.

5. A general association is apparent between lakes with elevated fish mercury concentrations and rock types which have good potential for sulphide mineralization. It is concluded that the pattern of fish mercury concentrations is broadly controlled by the bedrock of the region.

6. Smelting operations appear to have little or no general impact on mercury concentrations in fish. While there are abundant data to indicate that atmospheric fallout from the Hudson Bay Mining and Smelting Company complex at Flin Flon produces locally-elevated mercury concentrations in soil, snow, and vegetation, this impact appears to be limited to an area within a few tens of kilometers of the smelter. Fish from most lakes in the Flin Flon region contain below-average concentrations of mercury. Mercury data are not available for the area around Thompson but the distribution of other metals suggests that contamination from the smelter there is also limited to a radius of about 30 km. Because the ores processed at Thompson probably contain less mercury than those at Flin Flon, the overall impact of this smelter on general mercury levels is probably also smaller.

7. Active and inactive gold and base metal mining operations have had no identifiable effect on the mercury concentrations in fish in nearby lakes. None of the lakes with elevated fish mercury concentrations receive drainage from mine operations.

8. Water bodies which have experienced raised water levels due to impoundment or increased flow exhibit highly elevated fish mercury concentrations. This is especially notable along the Churchill Diversion-Burntwood River system but elevated concentrations in other impounded water bodies in northern Manitoba are documented in this report. It is concluded that many of the most-elevated fish mercury concentrations may be attributed to impoundment. A general relationship between age of impoundment and average fish mercury concentration was observed, in agreement with the conclusions reached in other studies that concentrations in impounded water bodies decline with time.

9. Studies in other regions have suggested that increasing water acidity may cause increased uptake of mercury by fish. In northern Manitoba, however, most lakes of significant size are not acidic and no relationship between water pH and fish mercury concentrations was found.

10. The mercury concentrations of lake sediments in the region do not appear to be elevated, although very high values occur

locally at sites which receive drainage from mine tailings. No relationship between mercury concentrations in lake sediments and fish could be demonstrated.

11. Forest fire may be an important factor in mobilizing mercury contained in vegetation and soils. Existing data do not permit firm conclusions but preliminary analysis suggests that further investigation of the role of fire is warranted.

12. With the exception of impounded water bodies, the mercury contamination of lakes in north-central Manitoba is natural in origin and is sufficiently widespread to warrant continued study. Although the source of the mercury is clearly the bedrock of the region, particularly rocks containing sulphide minerals, existing data are insufficient to identify the pathways by which the mercury moves from the rock into the aquatic ecosystem or the factors which govern its uptake by fish. To clarify these, this report recommends data collection programmes to determine the distribution of mercury in soils, vegetation, and snowmelt; continued monitoring of fish in selected lakes from all concentration categories; and special monitoring of lakes near forest fires, and water bodies which may be impounded in the future.

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ACKNOWLEDGEMENTS

The co-operation of a great many people is required in the preparation of a report such as this. Appreciation is expressed to all people listed in Appendix I for providing much of the data on which the report is based. Mr. Weldon Hiebert drafted all the maps and diagrams with his usual care and efficiency. Thanks are also due to Miss Brenda Hutt for her accuracy and forebearance in typesetting the text and especially the tables. Any errors in data presentation or interpretation are, of course, the responsibility of the authors.

A NOTE ON UNITS

The units commonly used to report mercury concentration data vary with the component of the environment. For example, lake sediment concentrations are usually expressed in parts per billion, fish concentrations in ug/g, rock concentrations in parts per million and surface water concentrations in micrograms per litre. A common unit such as parts per billion might be desirable in a report such as this (and for mercury studies in general) but because most researchers are familiar with concentrations expressed in the normal units for their own field of study, standard reporting units for each environmental component have been retained in this report. These units are defined below:

ppm = parts per million
ppb = parts per billion
ug/g = micrograms per gram
ug/L = micrograms per litre
1 gram = 1 x 10^6 ug
1 ug/g = 1 ppm
1 ug/L = 1 ppb
1 ug/kg = 1 ppb

PART ONE

INTRODUCTION

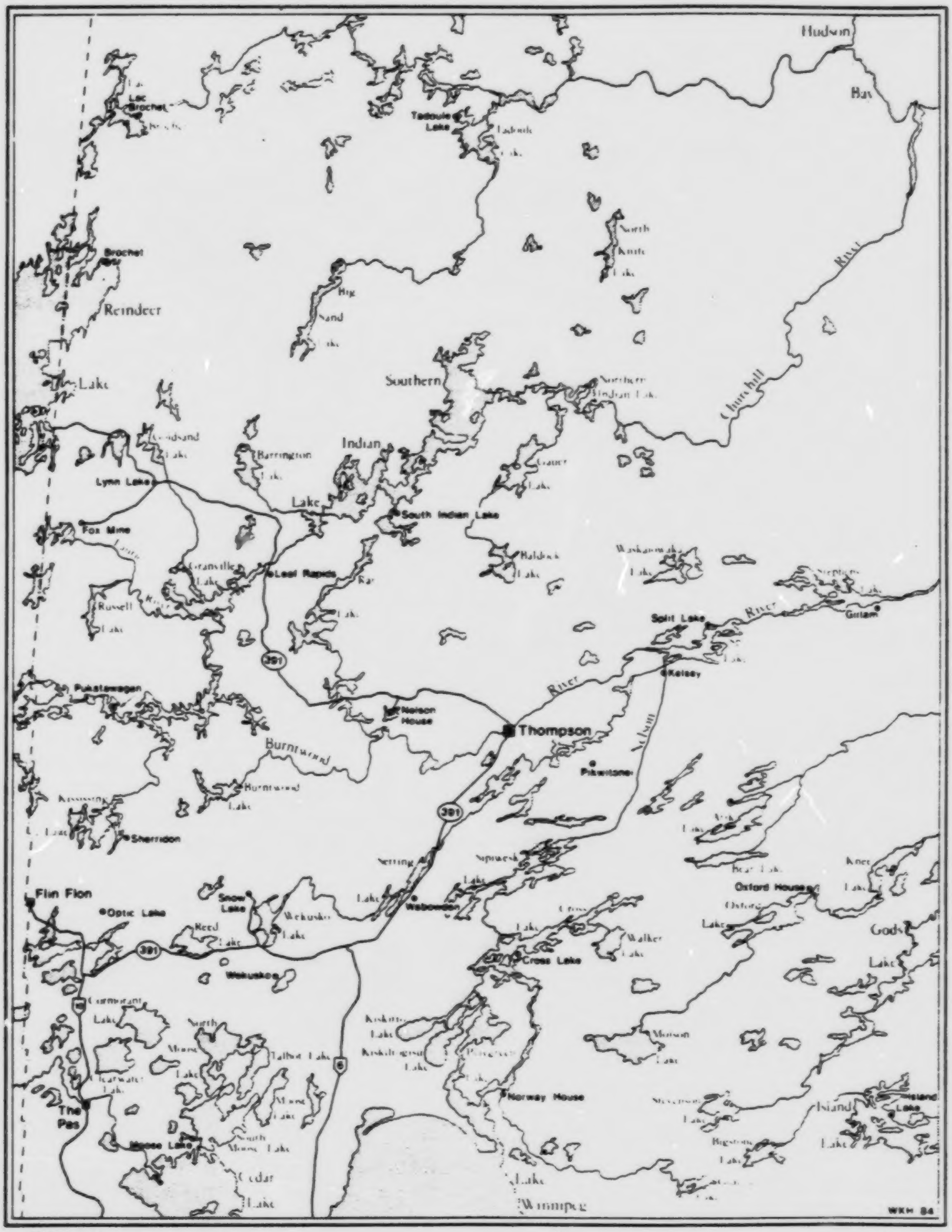
1.1 - OBJECTIVES

Evidence of elevated fish mercury levels in the water systems affected by the diversion of the Churchill River led to the signing of the Canada-Manitoba Agreement on the Study and Monitoring of Mercury in the Churchill River Diversion in March, 1983. High mercury levels measured in fish in other lakes, however, suggest that the problem of mercury contamination is not confined to the diversion route but occurs throughout a broad area of north-central Manitoba. Investigation of potential sources of mercury contamination was assigned to Project 1.2 of the Canada-Manitoba Agreement and forms the overall objective of this report. Specific objectives specified under the terms of contract are as follows:

1. To compile all available information on mercury in all sectors of the environment within the study region as defined in Part 1.2;

2. To compile information on known and potential point and non-point sources of mercury within the study region;

3. To evaluate the relative importance of each potential source; and


Hudson
Bay
Lac Brochet
Tadoule Lake
North Knife Lake
Brochet
Reindeer Lake
Big Sand Lake
Southern Indian Lake
Northern Indian Lake
Churchill River
Lynn Lake
Barrington Lake
South Indian Lake
Fox Mine
Granville Lake
Leaf Rapids
Rat Lake
Baldock Lake
Waskaiowaka Lake
Stephens Lake
Gillam
Russell Lake
Split Lake
Kelsey
Nelson House
Thompson
Pukatawagan
Burntwood
Burntwood Lake
Pikwitonei
Nelson River
Kississing Lake
Sherridon
Setting Lake
Sipiwesk Lake
Oxford House
Oxford Lake
Knee Lake
Flin Flon
Optic Lake
Snow Lake
Wekusko Lake
Wabowden
Reed Lake
Cross Lake
Walker Lake
Gods Lake
Wekusko
Cormorant Lake
North Moose Lake
Talbot Lake
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WKH 84

Figure 1: Study region.

4. To identify gaps in existing information and recommend additional work which might remove some of the gaps.

1.2 - STUDY REGION

The principal study region (Figure 1) specified by the contract is

> that part of northern Manitoba which is part of, or immediately adjacent to Southern Indian Lake and the Rat-Burntwood Rivers. For purposes of background information, the limits of the study area are approximately as follows: to the north, the Missi Falls outlet on Southern Indian Lake; to the west, upstream on the Churchill River, up to and including Reindeer Lake; to the south, upstream on the Nelson River, up to and including Sipiwesk Lake; and to the east, downstream on the Nelson River, up to and including Split Lake.

1.3 - GENERAL CONSIDERATIONS AND ORGANIZATION OF THE REPORT

Mercury is a naturally-occurring element of the earth's lithosphere that may be expected to occur in the environment in concentrations related to its abundance in the local bedrock. Mercury may also be "imported" into a region by natural or human agencies, producing levels which exceed those due to purely local sources. Regardless of the origin of the mercury within a region, various processes may concentrate or redistribute the element, creating new patterns of abundance. Finally, the actual levels in the biosphere (which are ultimately of greatest interest to man) are affected by factors which influence the type of mercury compound, its availability to organisms, and the way it becomes concentrated in the food chain.

The organization of the report reflects these system components. In Part 2, all available data on mercury levels within the study area are summarized. In Part 3 potential sources of mercury which are internal or external to the study region are evaluated. Related environmental factors which may be relevant to the distribution of fish mercury levels are discussed in Part 4.

Several excellent reviews of environmental mercury in Canada are available (e.g. Sherbin, 1979; National Research Council, 1979) and no attempt has been made to duplicate them here. Nevertheless, the concepts of "background" and "elevated" mercury levels are relative and can only be assessed with reference to levels reported elsewhere. Consequently, selected additional data are presented whenever possible to place the data for the study region in a more general context.

PART TWO

ENVIRONMENTAL MERCURY LEVELS

2.1 - BEDROCK

"Normal" or background mercury concentrations for a variety of rocks similar to those found in the study region are listed in Table 1 (Jonassen and Boyle, 1972) and concentrations in 58 rock samples from the Rice Lake region in the Superior Province of southeastern Manitoba are given in Table 2 (Weber and Stephenson, 1973). Many of the Rice Lake samples exhibit mercury levels which are elevated in comparison with the normal range, particularly gabbro, metavolcanic and volcanogenic metasediments. While some caution must be exercised in extrapolating from the Superior to the Churchill Province, rocks of similar type and origin are found in the study region and the Rice Lake data are considered to provide the best guidelines for general mercury levels in the bedrock of the study region.

Granite and granodiorite intrusions which are common in the study region probably have relatively small mercury contents. Much more important are the volcanic and related volcano-sediments (arkose, greywacke) of the Lynn Lake and Flin Flon Greenstone Belts and adjacent Southern Indian Lake and Kisseynew Sedimentary Gneiss Belts (Figure 2). If the Rice Lake values are representative, elevated mercury concentrations may be expected to be widespread throughout the study region.

TABLE 1

Average Mercury Contents of Some Common Rock Types
(after Jonasson and Boyle, 1972)

ROCK TYPE	MEAN (ppm)	RANGE (ppm)
Igneous		
Intermediate extrusives (andesite)	0.066	0.020-0.200
Acidic intrusive (rhyolite)	0.062	0.002-0.200
Acidic instrusives (granite, grnodiorite)	0.062	0.007-0.200
Intermediate instrusives (diorite)	0.038	0.013-0.064
Basic extrusives (basalt)	0.020	0.005-0.040
Basic intrusives (gabbro)	0.028	0.005-0.084
Metamorphic		
Schists	0.100	0.010-1.000
Quartzite	0.053	0.010-0.100
Amphibolites	0.050	0.030-0.090
Gneiss	0.050	0.025-0.100

TABLE 2

Mercury Content of Rocks in the Rice Lake Region,
Manitoba (Weber and Stephenson, 1973)

ROCK TYPE	n	MEAN (ppm)	RANGE (ppm)
Ultramafic	3	0.061	0.086-0.268
Meta-arkose, meta-conglomerate	5	0.742	0.289-1.420
Granodiorite, quartz monzonite	6	0.108	0.050-0.161
Paragneiss	2	0.045	0.044-0.046
Granodiorite, tonalite gneiss	4	0.095	0.053-0.161
Metasedimentary gneiss and schist	3	0.096	0.083-0.119
Metagreywacke, meta-argillite	4	0.698	0.205-1.625
Metadiabase	5	0.683	0.241-1.525
Quartz-feldspar porphyry	5	1.200	0.950-1.430
Quartz diorite	5	1.409	0.475-2.125
Dacite metavolcanics	6	0.823	0.329-2.000
Metagabbro	5	6.300	2.375-11.125
Metabasalt	5	0.451	0.161-0.825

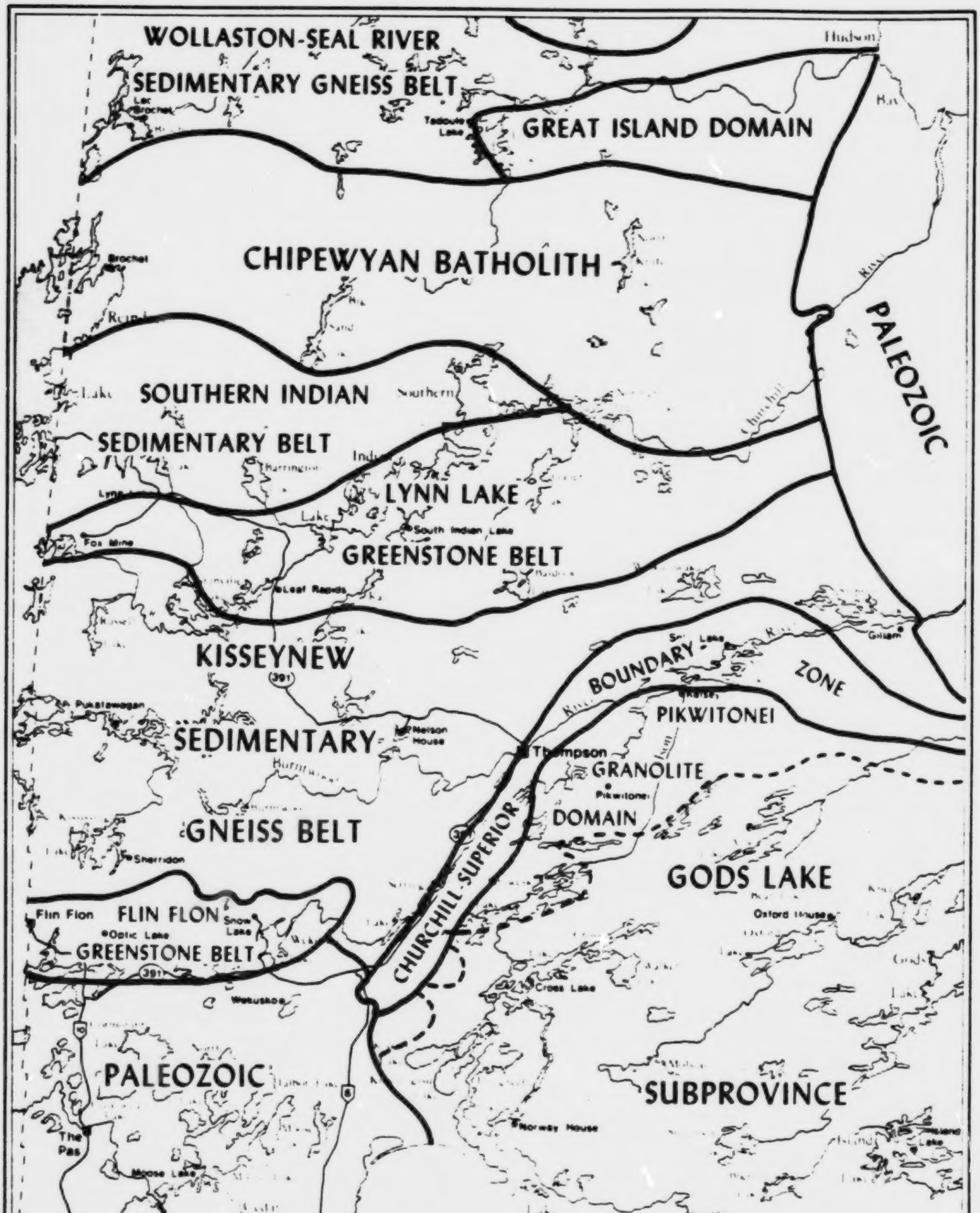

WOLLASTON-SEAL RIVER
SEDIMENTARY GNEISS BELT
GREAT ISLAND DOMAIN
CHIPEWYAN BATHOLITH
PALEOZOIC
SOUTHERN INDIAN
SEDIMENTARY BELT
LYNN LAKE
GREENSTONE BELT
KISSEYNEW
SEDIMENTARY
GNEISS BELT
BOUNDARY ZONE
PIKWITONEI
GRANOLITE
DOMAIN
CHURCHILL-SUPERIOR
GODS LAKE
SUBPROVINCE
FLIN FLON
GREENSTONE BELT
PALEOZOIC
Flin Flon
Thompson
Norway House
Cross Lake
Oxford House
Wekusko
Snow Lake
Sherridon
Pukatawagan
Nelson House
Leaf Rapids
South Indian Lake
Fox Mine
Brochet
Lac Brochet
Tadoule Lake
Gillam
Moose Lake
The Pas
Pikwitonei
Optic Lake
Hudson
WKH 84

Figure 2: Geologic belts within study region.

The most significant aspect of the geology of the study region is the massive sulphide mineralization which occurs in conjunction with the volcanogenic rocks. The association of mercury with massive sulphide ores is well-known, particularly in volcanogenic ore bodies where the mercury is preferentially associated with zinc sulphide (sphalerite) compared with sulphides of lead and copper (Jonasson and Sangster, 1974; Table 3). The mercury contents of eight sulphide ore bodies in the study region are listed in Table 4, with data for other Canadian mines for comparison in Table 5. From this data, Jonasson and Sangster (1974) concluded

> The Flin Flon-Snow Lake-Lynn Lake volcanogenic deposits...contain sphalerite of higher mercury content (up to 450 ppm) than similar deposits elsewhere in the Shield. (p. 313)

This conclusion is reinforced by the data in Table 6 which shows that the mercury contents of ore heads and tailings are much higher in Manitoba than at most other Canadian mines.

Mercury is also commonly associated with hypogene gold, although in much lower concentrations (less than 1 ppm) than in massive sulphides (Boyle, 1979). However, where the gold is associated with greenstones and volcanics containing sulphides, mercury concentrations of up to 25 ppm have been observed (Boyle,1979). Weber and Stephenson (1973) found patterns of mercury and gold concentrations to be virtually identical in the rocks of southeastern Manitoba.

TABLE 3

Mercury Content of Selected Common Sulphide Ore Minerals (after Jonasson and Boyle, 1972)

MINERAL	COMPOSITION	NORMAL RANGE (ppm)
Sphalerite	ZnS	0.1-200
Pyrite	FeS_2	0.1-100
Galena	PbS	0.04-70
Chalcopyrite	$CuFeS_2$	0.1-40
Chalcocite	Cu_2S	0.1-25
Pyrrhotite	Fe*S	0.1-5
Arsenopyrite	FeAsS	0.1-3

TABLE 4

Mercury Content of Some Volcanogenic Sulphide Ore Concentrates in Manitoba

NAME OF OCCURRENCE	ROCK TYPE	ORE TYPE	n	MEAN (ppm)	RANGE (ppm)	SOURCE
Osborne Lake	meta-sed.	Cu, Zn	2	238	121-452	1
Fox Lake	meta-sed.	Cu-Zn	2	174	110-238	1
Ruttan Lake	meta-sed.	Cu-Zn, Pb	2	135	130-140	1
Chisel Lake	meta-sed.	Pb-Zn, Cu	3	106	40-193	1
Lynn Lake	-	Zn	1	89	-	2
Flin Flon	meta-volc.	Cu-Zn	2	43	32-54	1
Schist Lake	meta-volc.	Cu-Zn	3	41	11-86	1
Sherridan	meta-volc.	Cu-Zn	5	18	3-40	1

Sources: 1 - Jonasson and Sangster, 1974
2 - Maclatchy and Jonasson, 1974

TABLE 5

Mercury Content of Some Volcanogenic Sulphide Ore Concentrates in other Canadian Locations

NAME OF OCCURRENCE	ROCK TYPE	ORE TYPE	n	MEAN (ppm)	RANGE (ppm)	SOURCE
Indian Mtn. Lake, NWT	meta-rhyol.	Pb-Zn	3	229	90-312	1
N. Contact Lake, Sask.	meta-sed.	Cu-Zn	1	200	-	1
Errington, Ont.	tuff	Cu-Zn, Pb	3	161	140-194	1
Manitou-Barvue, Que.	tuff	Cu-Pb, Zn	5	156	152-166	1
Key Anacon, N.B.	meta-volc.	Pb-Zn	2	116	70-161	1
Faro, Y.T.	meta-sed.	Pb-Zn, Ag	2	114	80-157	1
Sturgeon Lake, Ont.	volc.	Cu-Zn	1	83	-	1
Western Nuclear, Sask.	meta-volc.	Pb-Zn, Cu	1	73	-	1
Weedon, Que.	meta-tuff	Cu-Zn	2	61	26-96	1
Canadian Jamieson Ont.	rhyolite	Cu-Zn	3	60	40-70	1
Wedge, N.B.	meta-volc.	Pb-Zn	1	59	-	1
Mattabi, Ont.	tuff	Cu-Zn	6	53	27-100	1
New Calumet, Que.	meta-sed.	Pb-Zn, Ag	5	53	31-125	1
Kam-Kotia, Ont.	andesite	Cu-Zn	1	32	-	1
Seneca, B.C.	andesite	Pb-Zn, Cu	2	28	15-42	1
Quebec - all sulpnides			349	1.09	-	2
Quebec - zinc bearing sulphides			155	2.38	-	2
Quebec - non-zinc-bearing sulpnides			194	0.07	-	2

Sources 1 - Jonasson and Sangster, 1974
2 - Sears, 1970

TABLE 6

Average Mercury Contents (ppm) of Ore Heads and Tailings from Some Canadian Mines (Maclatchy and Jonasson, 1974

MINE	TYPE	HEADS	TAILINGS
Hudson Bay Mining and Smelting, Man.	Zn	24.9	1.20
Fox Lake, Manitoba	Zn	8.7	3.60
New Calumet, Quebec	Zn	7.5	0.56
Lac Dufault, Quebec	Cu, Zn	5.81	6.67
Manitou-Barvue, Quebec	Zn	4.65	1.27
Coast Copper Mines, B.C.	Cu	3.5	0.07
Normetal, Quebec	Cu, Zn	2.83	0.46
Coniagas	Zn, Pb	1.76	0.28
Mattagami Lake, Quebec	Zn, Cu	0.41	0.29

In summary, mercury is unusually abundant and widespread in the bedrock of the study region; indeed it may be one of the most naturally mercury-enriched regions of Canada outside of areas with known cinnabar mercury mineralization.

2.2 - SOILS

The mercury content of soils has been determined at 83 sites near the shorelines of Southern Indian Lake and lakes along the diversion route (Bodaly et al., 1984); for the surface organic layers at eleven sites in the vicinity of Flin Flon (McEachern and Phillips, 1983; see Section 2.3), and for several sulphide mineralized zones (Bradshaw, 1973). The results of these studies are summarized in Tables 7 and 8; Tables 9 and 10 provide data for other Canadian soils.

The mercury content of the Southern Indian Lake soils, while higher than elsewhere in Manitoba, appears typical of podzolic soils from other Canadian locations, both in terms of mean and range. The higher concentrations in the surface organic layer and A horizons at Southern Indian Lake are also typical; at the nine sites listed in Table 10 for which mercury concentrations in surface leaf and humus horizons were determined, concentrations in that layer were two to ten times greater than in non-organic lower horizons and McKeague and Wolynetz (1980) reported Ae/C and B/C ratios of 2.2 and 1.5 respectively for podzol soils. McKeague and Kloosterman (1974) found significant correlations ($p = 0.01$) between mercury concentration and organic carbon for several horizons in gleysolic and podzolic soils.

TABLE 7

Mean and ranges of Mercury Concentration (ppm) in Soil Horizons Within Study Region

HORIZON	n	SOUTHERN INDIAN LAKE REGION	FLIN FLON REGION
Moss, Litter, Humus	83	0.095 (<0.005-0.290)	
A	47	0.090 (<0.005-0.220)	
B	60	0.041 (<0.005-0.180)	
Surface organic layer -1980	11		6.59 (0.37-24.0)
Surface organic layer -1981	11		9.80 (0.43-22.0)

TABLE 8

Soil Mercury Concentration (ppm**) Over Sulphide Ore Bodies in the Flin Flon Area (Bradshaw, 1973)

LOCATION	HORIZON	STATION NUMBER						
		1	2	3	4	5	6	7
Keg Lake	A1/A2	7	1	356*	9	8	1	-
	A3	6	7	164	1	1	-	-
	A3/B	1	15	85	4	1	1	-
Pitching Lake	A1	32*	15	-	1	110	-	-
	A3/B	27	14	4	2	68	-	-
	C	3	1	1	1	-	-	-
Schist Lake	A1	125*	-	-	82	34	85	-
	A2	138	-	36	40	18	-	3
	A3	20	-	20	18	22	-	17
	A3/B	21	42	83	-	-	-	-
Shupe Lake	A2	5	3	21*	17	13	-	-
	A3	10	-	24	30	11	-	-
	B	-	21	-	-	-	-	-

* - indicates station closest to known ore body
** - see text for discussion of these units

TABLE 9
Mercury Concentrations (ppm) in Some Soils of Southern Manitoba (Mills and Zwarich, 1975)

TYPE OF LOCATION	n	MEAN	RANGE
Agricultural - A horizon	16	0.033	0.02 -0.053
Agricultural - C horizon	16	0.039	0.015-0.076
Highway - adjacent ditch	6	0.024	0.022-0.026
Highway - edge of field	6	0.025	0.018-0.032
Highway - 50 m away	6	0.021	0.016-0.025
Highway - 100 m away	6	0.025	0.017-0.033
Winnipeg urban area	11	0.033	0.023-0.040

TABLE 10
Miscellaneous Data on Soil Mercury Concentration (ppm) in some Canadian Soils

SOIL TYPE	HORIZON	n	MEAN	STANDARD DEVIATION
Canada-General (McKeague and Kloosterman, 1974)				
Podzolic	LH	2	0.165	-
	A	21	0.111	0.181
	B	27	0.137	0.134
	BC,C	29	0.102	0.306
Gleysolic	LH,H	3	0.177	-
	A	21	0.053	0.048
	B	17	0.029	0.019
	BC,C	23	0.023	0.016
Brunisolic	A	6	0.026	0.019
	B	14	0.038	0.031
	BC,C	15	0.036	0.026
Luvisolic	LH	1	0.120	-
	A	17	0.034	0.020
	B	12	0.066	0.054
	BC,C	13	0.038	0.017
Chernozemic	LH	3	0.086	-
	A	3	0.040	-
	B	3	0.040	-
Average - all Canadian soils		233	0.081	0.186
Cobalt, Ontario area (Boyle, 1966)				
Soil on till	A	-	0.200	-
	B	-	0.110	-
	C	-	0.200	-
Soil on sand and gravel	A	-	0.500	-
	A2	-	0.200	-
	B	-	0.300	-
	C	-	0.100	-
Soil on clay	A	-	0.200	-
	B or G	-	0.140	-
	C	-	0.110	-

The exceptional elevation of mercury in the surface organic layer in the vicinity of Flin Flon is related to fallout from the Hudson Bay Mining and Smelter operations at Flin Flon (see Part 3.3.3). Thus the mean for these samples is misleading since values are highly correlated with distance from the smelter. The lowest values, at a distance of about 40 km from the smelter, were considered by McEachern and Phillips (1983) to be background levels; but even at that distance, mercury concentrations were several times greater than those reported from the Southern Indian Lake region.

The paucity of soil samples is unfortunate in view of the apparently widespread elevated mercury content of the bedrock discussed in Part 2.1. Mercury halos in soils above massive sulphide ore bodies have been reported in a number of studies and in most cases such halos are sufficiently strong to enable mercury to be used as a pathfinder element in geochemical prospecting. For example, Friedrich and Hawkes (1966) found pronounced mercury anomalies in soils 15-60 m above massive volcanogenic sulphides. Mercury concentrations in the anomalies were up to ten times background levels but the anomalies were confined very closely to the soils immediately above the ore bodies (Figure 3). Note however the anomalous peak at Station 12 in the upper figure not associated with a known ore body. Hornbrook and Jonasson (1971) found mercury concentrations three times normal in soils overlying two sulphide gossans at Kaminak Lake, N.W.T. Significantly, the till in that area was frozen and Hornbrook and Jonasson concluded

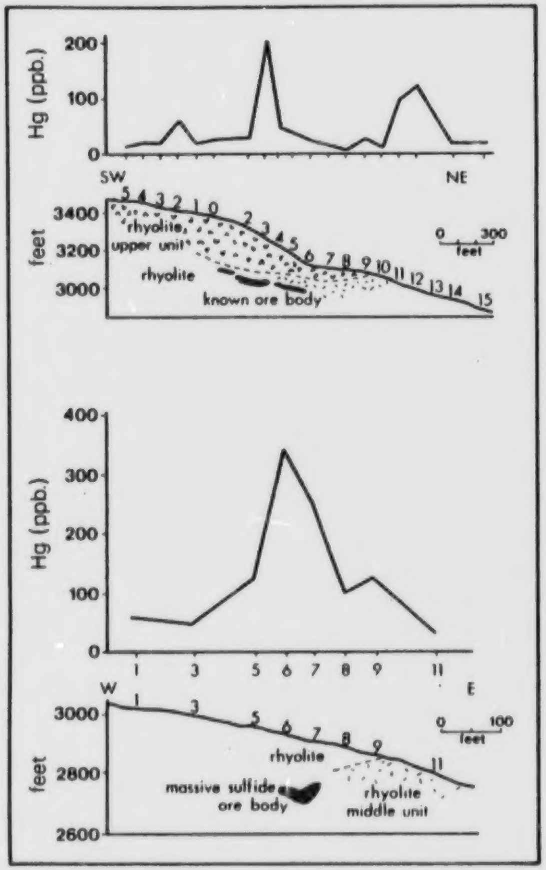


Figure 3: Mercury anomalies in soils near sulphide ore bodies, West Shasta, California (redrawn from Friederich and Hawkes, 1966).

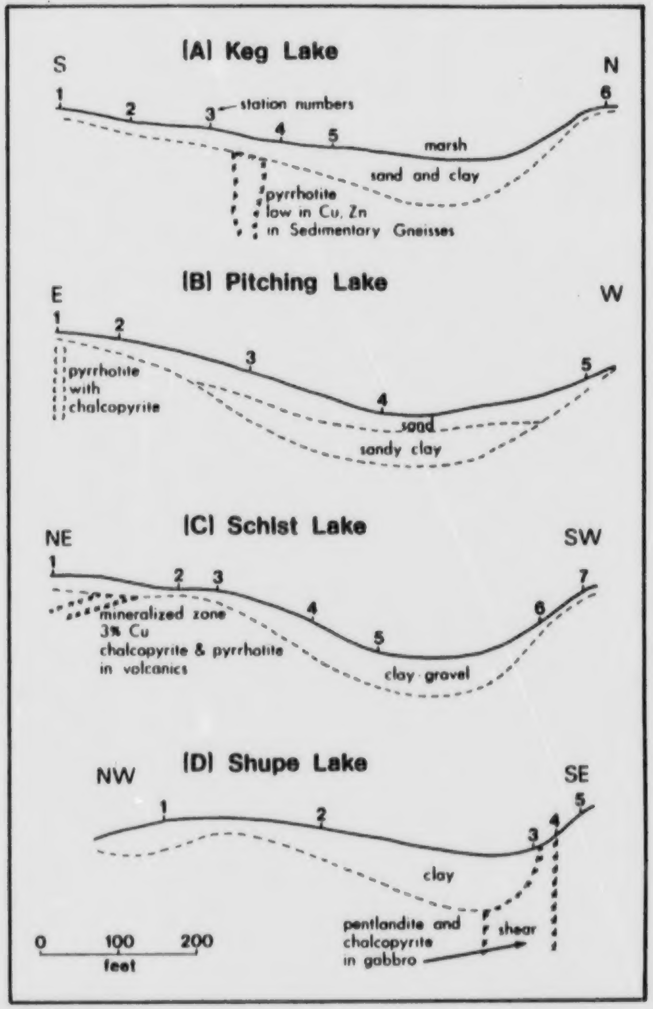

[A] Keg Lake
S
N
station numbers
marsh
sand and clay
pyrrhotite low in Cu, Zn in Sedimentary Gneisses
[B] Pitching Lake
E
W
pyrrhotite with chalcopyrite
sand
sandy clay
[C] Schist Lake
NE
SW
mineralized zone 3% Cu
chalcopyrite & pyrrhotite in volcanics
clay gravel
[D] Shupe Lake
NW
SE
clay
pentlandite and chalcopyrite in gabbro
shear
0 100 200 feet

Figure 4: Locations of soil mercury samples in the vicinity of sulphide ore bodies near Flin Flon (Table 8; redrawn from Bradshaw, 1971).

> ...mercury is penetrating extensive thickness of frozen till in the Kaminak Lake area producing a distribution directly related to the bedrock geology. (p. 5).

Bradshaw (1973) reported soil mercury concentrations for four transects across ore bodies in the Flin Flon area (Table 8 and Figure 4). As reported by Bradshaw (and in Table 8), these data appear suspect since maximum values are more than three orders of magnitude greater than the values for other Manitoba locations. A discrepancy of this magnitude does not seem to be explainable by their location near sulphide mineralization since the Flin Flon data are also three orders greater than values reported by Friedrich and Hawkes (1966) over volcanogenic sulphide ore bodies in California (see Figure 3). Thus it is concluded that parts per billion were inadvertently reported by Bradshaw as parts per million, and as a consequence, all values in Table 8 should be divided by 1000. Even allowing for this, however, the maximum values near ore bodies are elevated. Anomalies are apparent at Schist Lake and particularly at Keg Lake where a concentration of 0.356 ppm was recorded in the A horizon. Shupe Lake showed no anomaly, and at Pitching Lake an anomalous reading (Station 5, Table 8) was not associated with a known mineralized zone.

As was noted in Part 2.1, zones of potential sulphide mineralization are widespread in the study region and elevated soil mercury contents might be expected to show a similar distribution. Mercury anomalies due entirely to diffusion from a bedrock source are normally concentrated in the immediate vicinity of the ore body. Glacial processes, however, may redistribute this mercury-enriched material away from the bedrock source, producing secondary, more widespread anomalies and a distorted relationship between soil and bedrock mercury levels. Most glacial drift is of local origin, with the bulk composition reflecting the lithology within a few tens of kilometers, and the concentration of erratic materials usually declines rapidly away from the source. Nevertheless, fans of elevated soil mercury may be expected down-ice of areas of sulphide mineralization. Over most of the study region, ice flow was from the north (Figure 5); late-glacial flow from the northeast occurred in the area southeast of the Burntwood Interlobate Moraine. Thus, any redistribution of mercury due to glacial transport would be generally toward the south of bedrock sources. The strength and extent of these fans is frequently complex, however, depending on such factors as the mercury content of the source bedrock, the thickness and texture of the drift, local deviations in iceflow direction, etc. One further consequence of the dominant north-to-south direction of ice flow is that glacial transport of mercury-rich material into the study region is likely to have been insignificant. The lithology to the north is dominated by granite-quartz monzonite igneous intrusives of the Chipewayan Batholith, rocks which normally have low background mercury contents and little potential for sulphide mineralization (Tables 1 and 2). This conclusion is reinforced by the somewhat lower mercury levels in lake sediments in this region (see Part 2.4, Table 14).

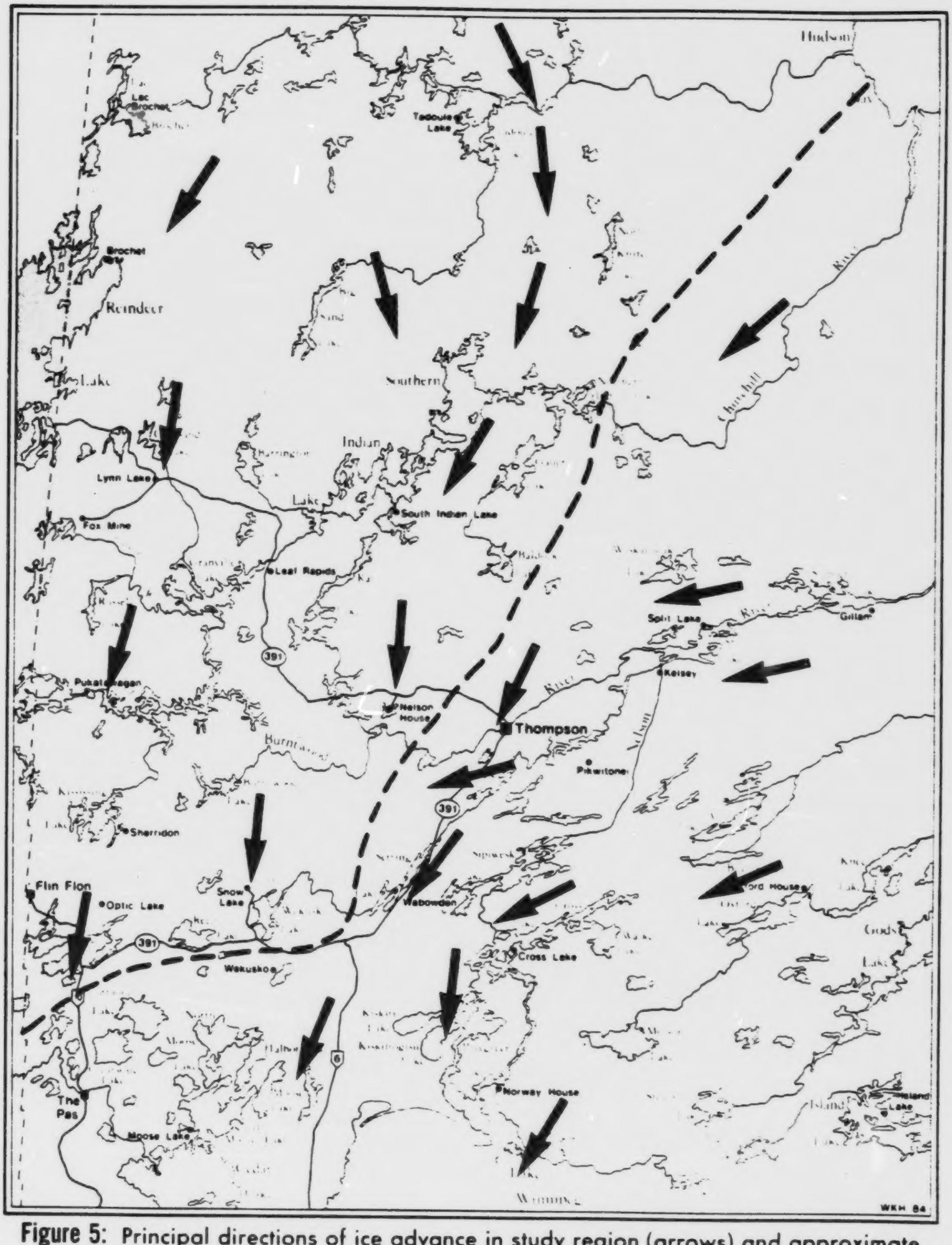

Hudson
Lac Brochet
Tadoule Lake
Brochet
Reindeer
Lake
Southern
Indian
Lake
Lynn Lake
Fox Mine
South Indian Lake
Leaf Rapids
Churchill
River
Split Lake
Gillam
Kelsey
391
Pukatawagan
Nelson House
Burntwood
Thompson
Pikwitonei
Nelson
Sherridon
Sipiwesk
Flin Flon
Optic Lake
Snow Lake
Wabowden
Cross Lake
Gods
Lake
Island
Lake
Wekusko
6
The Pas
Moose Lake
Norway House
Winnipeg
WKH 84

Figure 5: Principal directions of ice advance in study region (arrows) and approximate late-glacial contact between Labradorean and Keewatin Ice Sheets (dashed line) (after Manitoba Mineral Resources Division, 1981).

2.4 - VEGETATION

Mercury concentrations in blueberry (Vaccinium myrtilloides) plant components and the associated soil surface organic layer have been reported by Wotten (1979) and McEachern and Phillips (1983) for areas near the Hudson Bay Mining and Smelting complex at Flin Flon (Table 11) and for blueberry, rock cranberry (Vaccinium vitis-idaea) and common bearberry (Arctostaphylos uva-ursi) and bearberry in the same area by Shaw (1979) (Table 12). Mercury does not appear to accumulate in the fruit and occurs in only moderate amounts in the leaves, stems and roots, in general agreement with the observation by Erickson and Mortimer (1975) that plant uptake of mercury through root systems is relatively small.

As was noted in the previous section, mercury occurs in very high concentrations in the surface organic layer of the soils (Table 11). In all cases, the highest levels are at sites closest to the smelter and fall off rapidly within 20-40 km. The high correlation between mercury content and distance from the smelter clearly implicates atmospheric fallout from the smelter as the source. Other data support this conclusion (see Part 3.3.3).

No other measurements of mercury levels in vegetation within the study region are known.

TABLE 11

Mercury Concentration (ug/g) in Blueberry Plant Components and LFH, Flin Flon, 1979, 1980 & 1981.

Dispersion Sector	Distance* (km)	Site No.	Unwashed Berries			Leaves			Stems			Roots			LFH		
			1979	1980	1981	1979	1980	1981	1979	1980	1981	1979	1980	1981	1979	1980	1981
N-NE	2.0	11	-	0.03	0.03	-	0.21	0.21	-	0.15	0.16	-	0.08	0.48	-	7.80	17.80
	4.6	8	0.17	0.03	0.02	0.27	0.20	0.34	0.13	0.18	0.47	0.92	0.06	0.46	0.30	4.33	4.38
NE-E	3.2	10	-	0.04	0.02	-	0.30	0.12	-	0.24	0.10	-	0.20	0.07	-	5.36	6.45
E-SE	3.7	9	0.04	0.03	0.02	0.25	0.29	0.28	0.69	0.55	0.67	0.12	0.20	0.08	9.39	8.54	22.00
	5.1	1	<0.02	0.03	0.02	0.17	0.28	0.15	0.30	0.44	0.34	0.07	0.04	0.03	0.93	7.64	10.20
	8.9	2	0.08	0.03	0.03	0.17	0.20	0.18	0.17	0.20	0.21	0.03	0.04	0.04	1.87	2.06	12.40
	15.5	3	0.03	0.03	0.02	0.05	0.12	0.11	0.03	0.11	0.05	0.02	0.04	0.03	0.23	1.05	3.21
	38.9	4	0.02	0.03	0.02	0.03	0.06	0.04	0.04	0.04	0.02	0.04	<0.02	0.03	0.49	0.37	0.43
SE-S	2.5	7	0.16	0.04	0.02	0.41	0.33	0.57	0.88	1.18	5.16	0.68	0.07	0.03	10.86	10.00	17.80
	5.8	5	0.12	0.03	0.02	0.26	0.22	0.13	0.29	0.22	0.12	0.74	0.04	0.09	3.98	24.00	11.70
	19.6	6	0.05	0.03	0.02	0.11	0.07	0.04	0.20	0.09	0.07	0.05	0.04	0.05	1.06	1.36	1.43

* Distance from HBM&S complex

Data Sources: 1979 - Watten, 1979
1980, 1981 - McEachern and Phillips, 1983

TABLE 12

Mercury Concentration (ug/g) in Plant Components at Site SE of Flin Flon (Shaw, 1981)

PLANT	COMPONENT	MEAN CONCENTRATION (ug/g) LOUIS LAKE (2.4 km*)	MYSTIC CREEK (26.5*)	CRANBERRY PORTAGE (36.2*)
Blueberry	Berry	0.22	0.01	0.02
	Leaf	0.47	0.07	0.07
	Stem	1.09	0.09	0.26
Common bearberry	Berry	0.07	<0.01	<0.01
	Leaf	0.34	0.05	0.02
	Stem	1.45	0.05	0.03
Rock cranberry	Berry	0.46	0.03	0.02
	Leaf	0.66	0.07	0.04
	Stem	1.16	0.05	0.04

* - distance from HBM+S complex

2.4 - LAKE SEDIMENTS

Measurements of the mercury content in samples of lake sediments constitutes one of the largest mercury data bases within the study region. As part of the National Geochemical Reconnaissance Survey, lake sediment mercury content has been determined for approximately 8,100 sites in a 102,000 km^2 area of northern Manitoba. The 1:250,000 NTS maps covered by the surveys published to date are shown in Figure 6. Although most of the data are from the northernmost area of the province, one map, 64C, covers the Lynn Lake-Granville Lake area of the present study region (Geological Survey of Canada, 1983), and data for another section (covering NTS Maps 64B, F, and G) will become available in May-June, 1985 (E. Hornbrook, personal communication). Extensive statistical analysis of the data for the 1293 sample sites on Map 64C is beyond the scope of this report and should await publication of the survey results for Maps 64B, F, and G. For the purposes of this study, the data are summarized in Table 13. Comparable data from other areas surveyed in the National Geochemical Reconnaissance programme are given in Table 14. Lake sediment mercury concentrations for other lakes within or near the study region are given in Table 15 and Table 16 provides selected data from lakes in other regions of Canada.

Examination of Tables 13, 14 and 15 suggests that, with some notable exceptions, mercury concentrations in lake sediments within

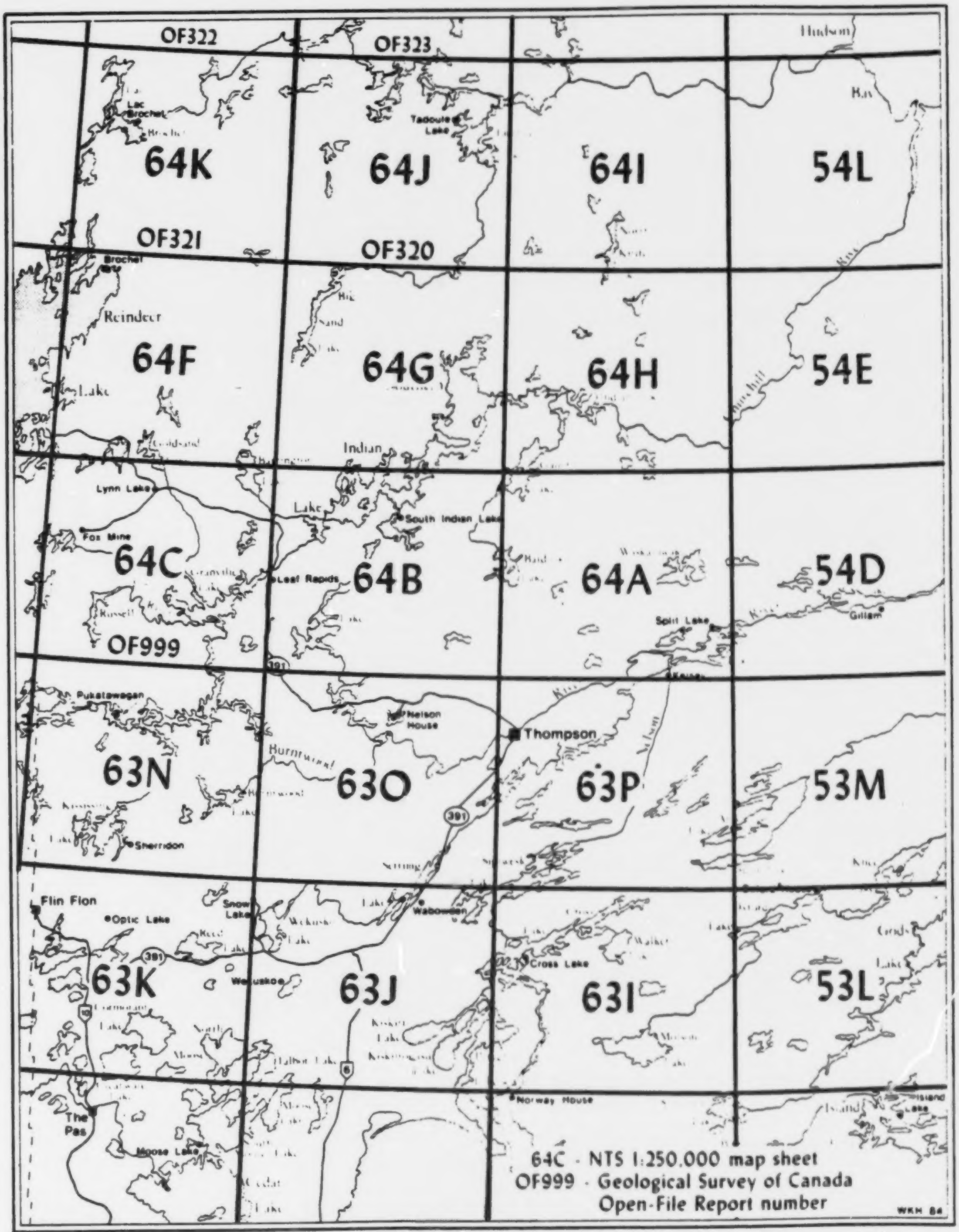

OF322
OF323
Hudson
Bay
Lac Brochet
Tadoule Lake
64K
64J
64I
54L
OF321
OF320
Brochet
Reindeer
Big Sand Lake
64F
64G
64H
54E
Lake
Churchill
River
Indian
Lynn Lake
Lake
South Indian Lake
Fox Mine
64C
Leaf Rapids
64B
64A
54D
Russell
Split Lake
Gillam
OF999
391
Pukatawagan
Nelson House
Thompson
Burntwood
63N
63O
63P
53M
Sherridon
Flin Flon
Optic Lake
Snow Lake
Wabowden
Cross Lake
63K
Wekusko
63J
63I
53L
Walker
Gods Lake
Moose
6
10
Norway House
Island Lake
The Pas
Moose Lake
Cedar Lake
64C - NTS 1:250,000 map sheet
OF999 - Geological Survey of Canada Open-File Report number
WKH 84

Figure 6: Coverage of study region by National Geochemical Reconnaissance Lake Sediment Surveys.

TABLE 13

Summary of Regional Lake Sediment Mercury Concentration (ppb, dry weight), NTS Map 64C (Geological Survey of Canada, 1983)

CONCENTRATION (ppb)	NO. OF SITES	PERCENT	CUMULATIVE NO. OF SITES	CUMULATIVE PERCENT
5-10	1	0.08	1	0.08
10-20	28	2.2	29	2.2
20-50	554	42.9	583	45.1
50-100	690	53.4	1273	98.5
100-200	18	1.4	1291	99.9
200-500	1	0.08	1292	99.9
500-1000	1	0.08	1293	100.0

Arithmetic Mean - 59
Standard Deviation - 33
Geometric Mean - 55
Log$_{10}$ Standard Deviation - 0.15
Minimum - 10
Median - 60
Maximum - 960

TABLE 14

Comparison of Mercury Concentrations (ppb, dry weight) of Lake Sediments in Other Geochmical Reconnaissance Survey Areas (Coker and Shilts, 1979)

REGION	NTS 1:250000 MAP SHEETS	n	ARITHMETIC MEAN (ppb)	S.D.	RANGE
Study Region	64C	1293	59	33	10-960
Northwestern Manitoba	64J, K, N, O	3661	43	22	5-190
North Shore, Lake Superior	52A, H; 42C-F, 41K, N	1816	133	58	20-710
Southeastern Ontario	31C, F	1250	94	54	10-580
All samples collected by Geological Survey in Shield to end of 1981		29174	62	-	-

TABLE 15

Miscellaneous Lake Sediment Mercury Concentrations (ppb, dryweight)
Within and Near Study Region

LAKE	MEAN (ppb)	S.D.	RANGE	SOURCE
+1. Hamell Lake	1160	-	-	1
+2. Hook Lake	530	-	-	1
+3. Cliff Lake	400	-	-	1
4. Thompson	90	-	-	1
5. Nesootao	60	-	-	1
6. Footprint Lake	80	-	-	2
7. Churchill River, Granville Lake	20	-	-	2
*8. Snake Lake near outlet	290	70	-	2
9. Cockeram Lake	80	30	-	2
10. Barrington River upstream of Apachuanau Lake	70	5	-	2
11. Rusty Lake near outlet	70	10	-	2
12. Southern Indian Lake at settlement	50	5	-	2
13. Split Lake near community	40	5	-	2
14. Apussigamasi Lake	40	-	-	2
15. Reed Lake	90	20	-	2
16. Wekusko Lake at Anderson Bay	70	17	-	2
*17. Wekusko Lake at Herb Bay	100	10	-	2
18. Upper Ospawagan Lake	40	10	-	2
19. Cross Lake at settlement	40	-	-	2
20. Nelson River at Norway House	30	5	-	2
21. Cedar Lake at Oleson Point	70	5	-	2
22. Moose Lake at settlement	50	5	-	2
23. Burntwood River, Thompson	20	-	-	2
*24. Schist Lake	240	60	-	2
25. Athapapuskow Lake at Baker's Narrows	150	350	-	2
26. Southern Indian Lake (n=28)	36	-	10-70	3
27. God's Lake near outlet (n=15)	30	12	<20-60	4
28. God's Lake near outlet (n=15)	<20	-	-	4
*29. God's Lake near tailings drainage	300	513	<20-1570	4
30. Sherlett Lake	320	130	<30-660	5
*31. Camp Lake	1140	690	420-2420	5
*32. Cold Lake at outfall of Camp Lake	370	400	50-1460	5
33. Cold Lake at inlet	460	270	30-790	5
34. Snow Lake at inlet	78	53	20-160	6
*35. Snow Lake near tailings drainage	68	44	<20-120	6
*36. Snow Lake downstream of tailings	110	46	30-170	6
37. Herblet Lake near outlet	<20	-	-	6
*38. Herblet Lake downstream of tailings drainage	50	72	20-190	6
*39. Herblet Lake near tailings	30	30	<20-120	6
40. Wekusko Lake - Goose Bay	50	-	-	7
41. Wekusko Lake - Berry Bay	80	-	-	7
*42. Wekusko Lake - Crowduck Bay	510	-	-	7
*43. Wekusko Lake - all samples	160	-	-	7
44. Anderson Lake	110	-	-	7

TABLE 15 Continued

LAKE	MEAN (ppb)	S.D.	RANGE	SOURCE
45. Granville Lake, 1981	34	-	-	8
46. Granville Lake, 1983	30	-	20-30	8
47. East Mynarski Lake	90	-	-	8
48. East Notigi Lake, June, 1981	121	-	117-124	8
49. East Notigi Lake, Aug., 1981	131	-	130-132	8
50. East Notigi Lake, Aug., 1983	60	-	50-70	8
51. West Notigi Lake, June, 1981	58	-	57-59	8
52. West Notigi Lake, Aug., 1981	57	-	55-60	8
53. West Notigi Lake, Aug., 1983	75	-	70-80	8
54. Rat Lake, June, 1981	77	-	76-78	8
55. Rat Lake, Aug., 1981	90	-	87-93	8
56. Rat Lake, Aug., 1983	30	-	-	8
57. Central Mynarski Lake, June, 1981	79	-	75-82	8
58. Central Mynarski Lake, Aug., 1981	110	-	108-111	8
59. Central Mynarski Lake, Aug., 1983	70	-	-	8
60. West Mynarski Lake, Aug., 1983	70	-	-	8
61. Southern Indian Lake, central Long Bay, June, 1981	56	-	54-57	8
62. Southern Indian lake, Long Bay shallows, Aug., 1983	30	-	-	8
63. Southern Indian Lake, Long Bay shallows, June, 1981	57	-	55-58	8
64. Southern Indian Lake, Long Bay shallows, Aug., 1981	64	-	62-65	8
65. Southern Indian Lake, Long Bay shallows, Aug., 1983	40	-	-	8
66. Southern Indian Lake, South Bay, June, 1981	51	-	50-52	8
67. Southern Indian Lake, South Bay Aug., 1981	60	-	-	8
68. Southern Indian Lake, South Bay Aug., 1983	30	-	-	8
69. Southern Indian Lake, Methyl Bay flooded area, Aug., 1983	70	-	-	8
70. Southern Indian Lake, Methyl Bay, offshore, Aug., 1983	40	-	-	8

Sources:
1. McFarlane and Franzin, 1980
2. Williamson, 1980
3. Bodaly et al., 1984
4. Beck, 1982
5. Beck, 1983
6. Beck, 1984
7. Brown et al., 1980
8. Jackson, 1984

+ - within 10 km of smelter at Flin Flon
* - receives drainage from active or inactive mine tailings storage

the study region are generally unremarkable when compared with data from other Canadian regions. In the area covered by NTS Map 64C, although mean and maximum values are greater than those for the area further north, only 20 of the 1293 sites (1.5%) exceeded 100 ppb and only two were greater than 200 ppb. Both of the latter were from ponds with sediments of high organic content and only one water body with an area greater than 1 km^2 exceeded 100 ppb. A similar distribution is implied by the data in Table 15 if allowance is made for the bias in sampling toward areas where high values are to be expected. Even with this bias, the modal class is the same as that for the larger sample and the median concentration is similar.

Most of the highly elevated values in Table 15 are for lakes or portions of lakes which receive drainage from mine tailings storage sites, or which are near the smelter at Flin Flon. In at least some of these lakes, the high concentrations occur within very restricted areas. For example, the very high values in God's Lake (No. 29 in Table 15) are strongly correlated with distance from the mouth of the stream draining the tailings pile from an inactive gold mine and fall off to low background values of less than 50 ppb within 0.5 km (Beck, 1982). The total area of God's Lake displaying metal enrichment was estimated by Beck to be only 0.05 km^2. Consequently, the relatively high mean and very high extreme concentrations for this lake are misleading.

Concentrations higher than the regional average also occur in several lakes affected by the Churchill Diversion. East Notigi Lake, for example, had concentrations of 120-130 ppb in 1981 and values of 90 ppb or more were observed on East Mynarski, Central Mynarski, and Rat Lakes. Footprint Lake (80 ppb) was also somewhat above average. However, concentrations on all these lakes were much lower in 1983 and all values for Southern Indian Lake, which experiences extensive flooding and shoreline erosion due to the diversion, were near or below the regional average.

Much higher lake sediment mercury concentrations have been reported for lakes along the Reindeer and Churchill Rivers in Saskatchewan (Table 17). A particularly noteworthy feature of these data is the relatively high concentrations at depth in the cores. Elevation of mercury concentration in the surface layer of lake sediments is normal and is attributable to increased global atmospheric flux due to industrialization in this century. At Southern indian Lake within the study region, a gradual increase in mercury concentrations upward through a 20-cm. core was reported by Brunskill (Bodaly et al., 1984), and the top-to-bottom difference of 2X observed there was consistent with results reported from other areas with no nearby source of industrial emission. No consistent upward increase is evident in the cores from the Reindeer and Churchill River systems, however. Although some surface enrichment can be demonstrated (compare mean and median values in Table 17), and is clearly evident in some lakes (e.g. Wapumon, Iskwatam, The Two Rivers), in only 9 of the 16 cores did the highest concentration

TABLE 16

Means and Ranges of Mercury Concentrations in Sediments of Selected Canadian Lakes

LAKE	n	MEAN ppb	RANGE ppb	SOURCE
Lake Ontario, all samples	248	651	32-2100	Thomas, 1972
Lake Ontario, inshore	126	335	32-1820	Thomas, 1972
Lake Huron, all samples	163	222	54-805	Thomas, 1973
Lake Huron, inshore	89	171	54-665	Thomas, 1973
Lake Erie	6	858	330-1430	Kemp and Thomas, 1976
Peg Lake, N.W.T.	-	80	28-194	Moore et al., 1979
Keg Lake, N.W.T.	-	47	13-120	Moore et al., 1979
*Meg Lake, N.W.T.	-	132	24-433	Moore et al., 1979
Great Slave Lake, N.W.T.	-	53	11-151	Moore et al., 1979

* receives waste from operating gold mine

TABLE 17

Lake Sediment Mercury Concentrations (ppb, dry weight) from Lakes on Reindeer and Churchill River Systems in Saskatchewan (Sawchyn, 1975)

LAKE	WATER DEPTH (m)	SAMPLE DEPTH IN CORE (cm)		
		0-2	2-5	5-15
Faford Lake - 1	12	190	160	150
Faford Lake - 2	7	170	160	100
Steephill Lake - 1	22	210	130	140
Steephill Lake - 2	6	230	970	170
Royal Lake - 1	6	170	150	220
Royal Lake - 2	12	50	70	100
The Two Rivers - 1	5	420	200	560
The Two Rivers - 2	17.5	220	110	120
Iskwatam - 1	12.5	480	150	230
Iskwatam - 2	8	470	150	140
Wintego - 1	6	160	110	70
Wintego - 2	14	130	170	70
Wapumon - 1	13.5	480	200	300
Wapumon - 2	9	340	280	110
Pita Lake - 1	16	210	330	160
Pita Lake - 2	10	120	130	370
Median		210	160	150
Mean		253	217	188

occur in the surface layer. Because a sedimentary unit represents a specific time period, an isolated layer with mercury content higher than layers above and below implies a distinct period of increased mercury influx as opposed to a gradual change due to the cumulative increase from industrial emission. Such layers appear at the 2-5 cm depth in several of the lakes (Steephill, Core 2; Wintego, Core 2; Pita Lake, Core 1). Sedimentation rates were not determined for these lakes but if the rates reported by Heckey (1975) for Reindeer Lake are representative, the upper (0-2 cm) layers were deposited within the past few decades, the intermediate (2-5 cm) layer within the past century, and the lower (5-15 cm) layers within the last 1000 years. Thus factors other than increased global atmospheric flux appear capable of producing temporal variations in lake sediment mercury concentration.

2.5 - WATER

Mercury concentrations measured in water samples taken at 26 lake and river locations in this study region (Figure 7) are summarized in Table 18. Measurements taken before 1974 have been excluded because of suspected contamination of the samples (Chacko, 1984, personal communication). As is typical of most other natural waters without major anthropogenic contamination, virtually all concentrations are at or near the normal detection level (0.02 but occasionally 0.05 ug/L). It should not be inferred, however, that these very low concentrations are unimportant. Water is the medium through which most uptake of mercury by fish occurs and prolonged exposure to very low mercury concentrations (at or near the detection limit) has been shown to result in elevated fish mercury levels, depending upon the form of the mercury, the exposure rate and duration, and the fish species (Waite, 1981). Nevertheless, the mercury levels observed in the surface waters of the study region reveal little about spatial or temporal patterns in mercury transport.

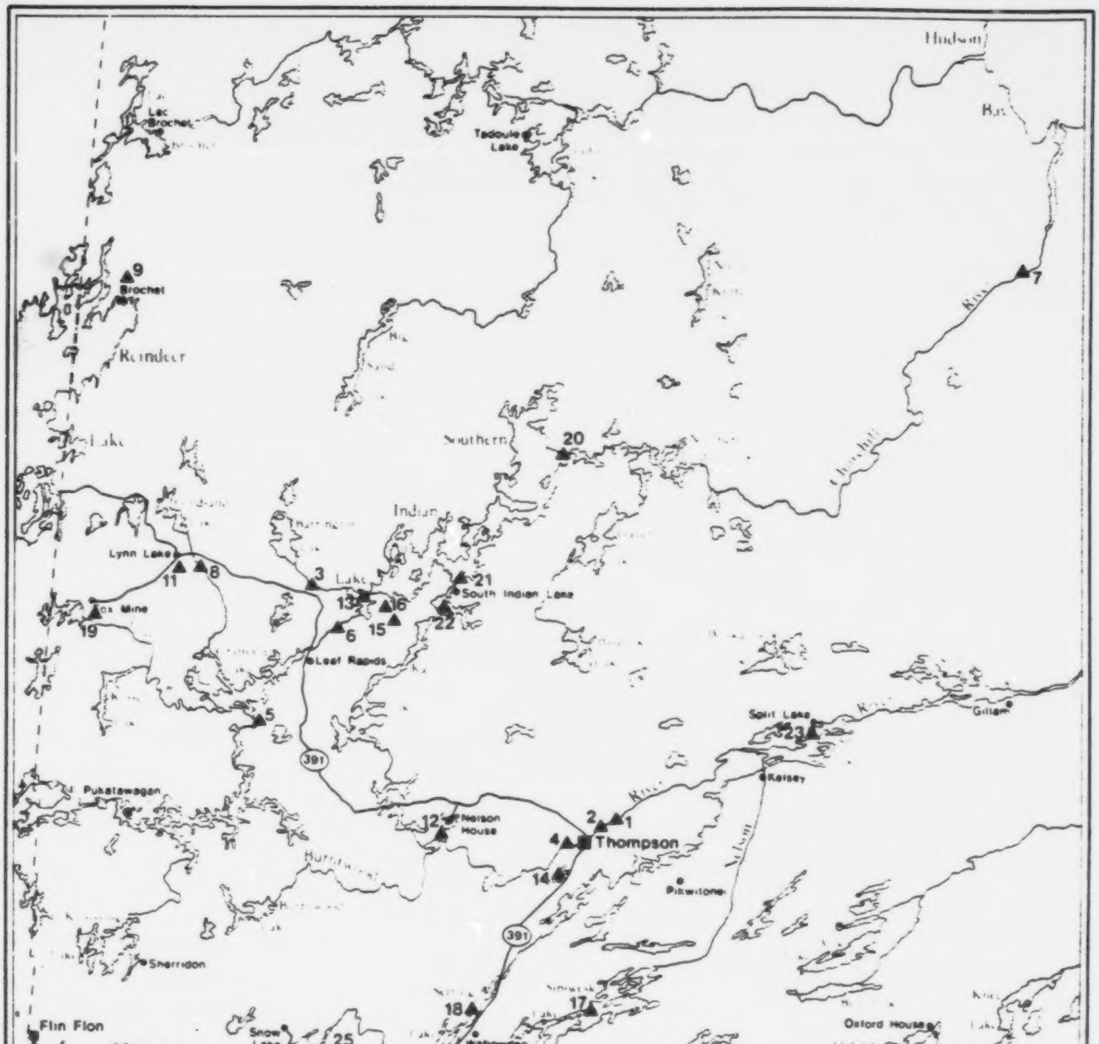

Hudson
Bay
Lac Brochet
Tadoule Lake
9
Brochet
Reindeer
Lake
7
River
Churchill
Southern
20
Indian
Lake
Lynn Lake
11
8
3
13
16
15
6
21
South Indian Lake
22
Fox Mine
19
Leaf Rapids
5
391
Split Lake
23
Gillam
Kelsey
River
2
1
4
Thompson
14
Nelson House
12
Pukatawagan
Burntwood
Pikwitonei
Nelson
Sherridon
Flin Flon
Optic Lake
Snow Lake
25
24
18
Wabowden
17
Oxford House
10
Cross Lake
Gods
Lake
Wekusko
10
6
The Pas
Moose Lake
Norway House
Island Lake
Lake
Winnipeg
WKH 84

Figure 7: Locations of water sampling stations listed in Table 18.

TABLE 18

Summary of Mercury Concentration Data for Lakes
and Rivers in Study Region

WATER	CONCENTRATION (ug/L)							
	n	<0.02	<0.05	0.02	0.03	0.04	0.05	Other
1. Apussigamasi Lake	18	16	2	-	-	-	-	-
2. Apussigamasi Lake	18	18	-	-	-	-	-	-
3. Barrington River	37	35	-	2	-	-	-	-
4. Burntwood River	43	38	2	2	-	1	-	-
5. Churchill River	35	33	-	2	-	-	-	-
6. Churchill River	9	7	-	-	2	-	-	-
7. Churchill River	4	3		-	-	1	-	-
8. Cockeram Lake	38	35	1	1	-	1	-	-
9. Cochrane River	10	6	-	3	1	-	-	-
10. Cross River	2	2	-	-	-	-	-	-
11. Eldon River	22	22	-	-	-	-	-	-
12. Footprint Lake	42	37	1	3	1	-	-	-
13. Opachuanau Lake	4	3	-	-	-	-	-	-
14. Upper Ospwagon Lake	42	37	1	1	1	-	1	0.06
15. Rusty Lake	31	29	-	1	-	1	-	-
16. Rusty Lake	38	33	1	-	2	-	1	0.08
17. Sipiwesk Lake	12	8	2	1	1	-	-	-
18. Setting Lake	7	5	2	-	-	-	-	-
19. Snake Lake	28	26	-	1	-	-	-	0.07
20. Southern Indian Lake	10	5	2	2	-	1	-	-
21. Southern Indian Lake	27	23	2	2	-	-	-	-
22. Southern Indian Lake	13	13	-	-	-	-	-	-
23. Split Lake	39	34	2	2	-	-	-	0.07
24. Wekusko Falls	10	10	-	-	-	-	-	-
25. Wekusko Lake	10	9	-	1	-	-	-	-

Note: Numbers below each concentration refer to the number of measurements with that concentration. Values listed under "other" are single measurements higher than 0.05 ug/L.

Data Sources: Environmental Management Division, 1984
Renewable Resources
NAQUADAT

2.6 - GROUNDWATER

No direct observations of the mercury content of groundwater in the study region are available. According to Jonasson and Boyle (1972), normal groundwater mercury levels elsewhere range from 0.01 to 0.10 ppb with a mean of about 0.05 ppb. Near sulphide deposits, groundwater and mine waters may contain up to 1000 ppb (National Research Council, 1979) but this value undoubtedly refers to sulphides of mercury (cinnabar) and cannot be applied to the base metal sulphides of the study region. Nevertheless, leaching of sulphide ores, concentrates and tailings piles occurs (see Part 3.3.2), and above-background mercury levels in groundwaters in the vicinity of these mineralized zones might be expected. Jonasson and Boyle (1972) however, suggested that mercury is not likely to be dispersed far from the source due to increased pH and adsorption onto clay, iron and organic matter. Consequently the importance of groundwater to the distribution of mercury within the study region remains conjectural.

2.7 - FISH

Because of their significance to man, mercury concentrations in fish have been determined for nearly 300 lakes within the study region. Analysis of this large body of data is frustrated, however, by many factors, among the most important of which are the following:

1. The size, frequency and type of sample varies greatly among the lakes, from small, single-year lake surveys to large samples obtained by lake survey and monitoring of commercial shipments over several years;

2. Mercury concentration is highly dependant upon species, generally being greatest in northern pike at the top of the aquatic food chain;

3. Within a given species, concentration is correlated with fish size (and thus age), making comparison of samples difficult unless size distribution is specified.

For these and other reasons, a detailed statistical treatment of this large data body is beyond the scope of this report. Instead, the emphasis here will be upon the spatial pattern displayed by the data, in keeping with the primary objective of the report to identify potential sources of contamination.

Appendix II summarizes available data on mercury concentrations in northern pike (Esox lucius), yellow pickerel (walleye) (Stizostedion vitreum) and lake whitefish (Coregonus clupeaformis) for 324 lakes in and near the study region, including 30 in Saskatchewan. Because northern pike usually have the highest mercury concentrations, the discussion which follows refers to that species unless otherwise indicated.

Histograms of lake mean mercury levels in northern pike from lake surveys and commercial monitoring reveal similar distributions for each survey type (Figures 8A, 8B). The data used to construct the histograms are mean concentrations by lake; concentrations in individual fish are obviously higher, exceeding 3.0 ug/g in some cases. The percentages of lakes with mean concentrations exceeding various health standards are given in Table 19.

TABLE 19

Percentages of Lakes with Mean Concentrations
Exceeding Various Health Standards

RECOMMENDED LEVEL FOR	CONCENTRATION ug/g	% OF LAKES EXCEEDING	
		COMMERCIAL MONITORING	LAKE SURVEY
(a) Those who consume large quantities	0.20	92	92
(b) Commercial fish (domestic sale)	0.50	38	44
(c) Commercial fish (export sale)	1.00	6	9

Clearly, lakes with fish mercury concentrations high enough to be of concern are the norm rather than the exception for the study region.

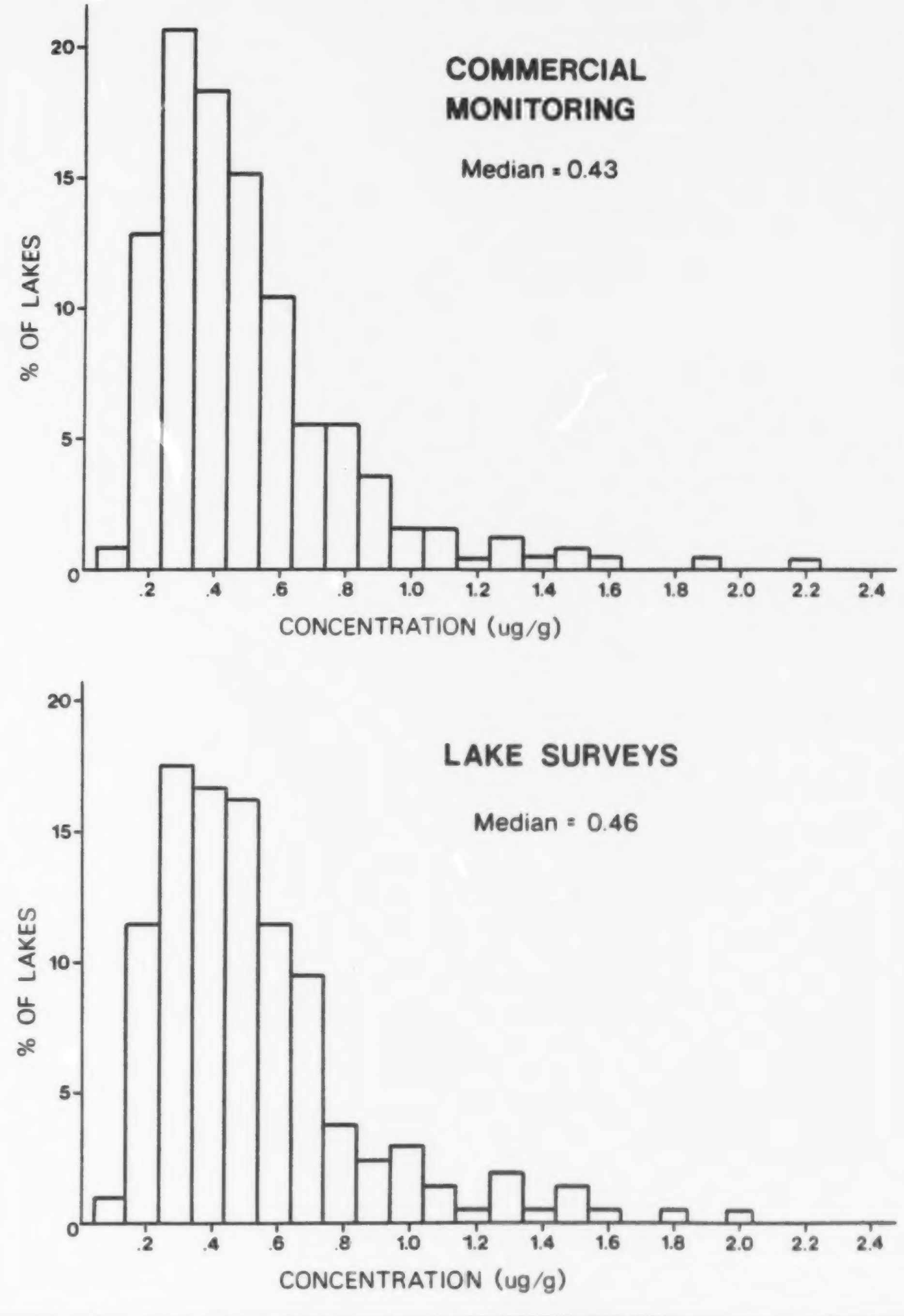


Figure 8: Distribution of lake mean mercury concentrations in northern pike.

Fish in Manitoba lakes appear to contain higher concentrations of mercury than those in northern Saskatchewan lakes. The distribution of mean concentrations in northern pike for commercial samples from 115 lakes in northern Saskatchewan (Murray, 1978) is compared with that of the study region in Figure 9. The median concentration (0.23 ug/g) in Saskatchewan is about half that for the study region and only 9% of the lakes there had means in excess of the standard for domestic sale. Significantly, all of these were located in that part of Saskatchewan immediately adjacent to the region of Manitoba discussed in this report.

The spatial patterns of mean northern pike mercury concentrations determined by lake survey and commercial monitoring are given in Figures 10 and 11. Most lakes with high concentrations (i.e. greater than 1.14) are located in a broad arc along the Southern Indian Lake - Rat River - Burntwood River - Nelson River waterway; a secondary concentration of lakes with means greater than 0.84 ug/g is apparent to the south and southwest of Lynn Lake. Lakes in the southern part of the study region are generally in the lower mean concentration categories, particularly in the areas within 150 km of Flin Flon and to the east of the Nelson River; the majority of the lakes in the lowest category are located in these areas and most others are below the median for the study region as a whole (Table 20). The relatively low concentrations near Flin Flon are significant since the smelter there is a potentially large source of contamination. The area to the east of the Nelson River is within the Superior Province whereas virtually all of the other lakes are within the Churchill Province.

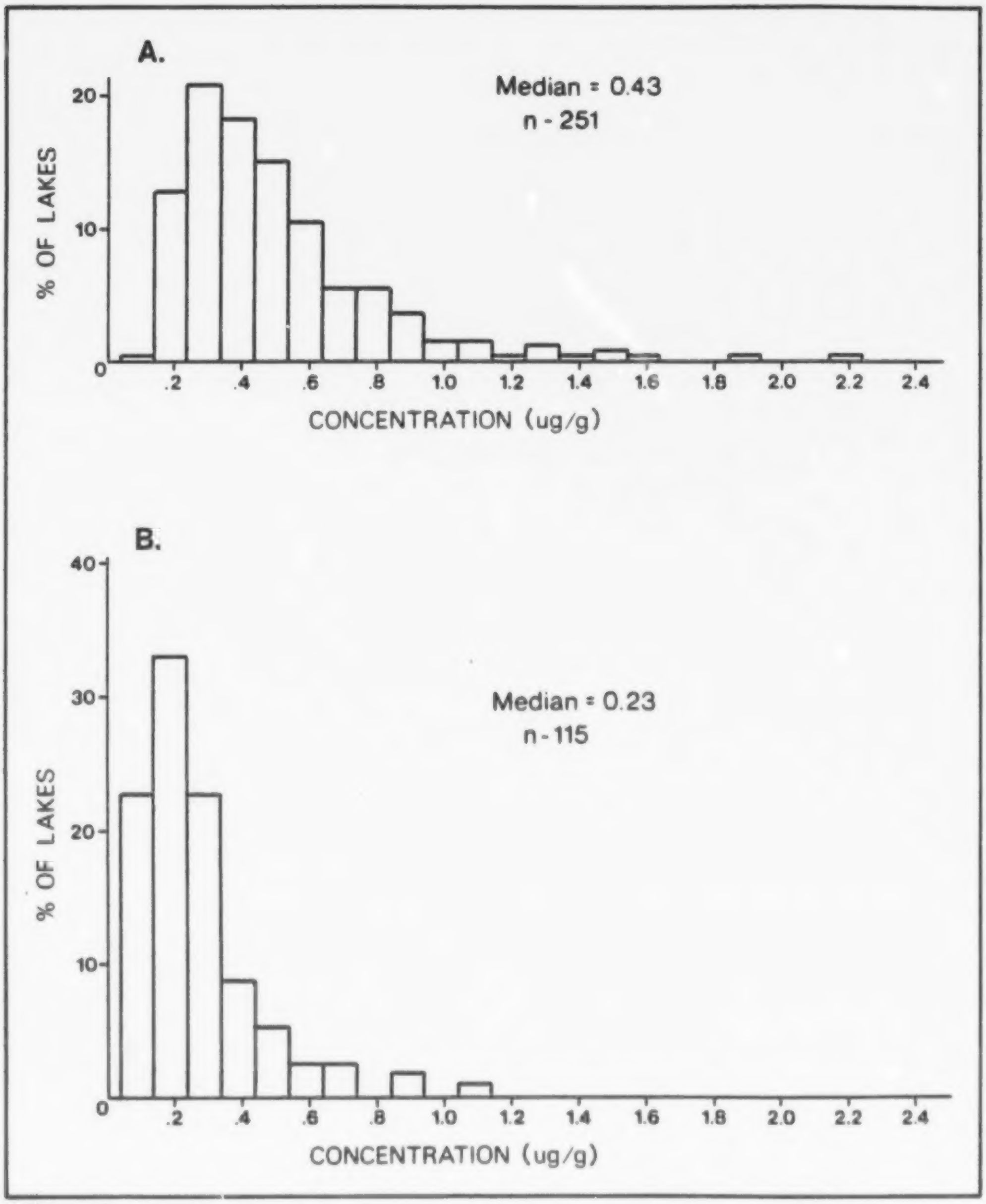
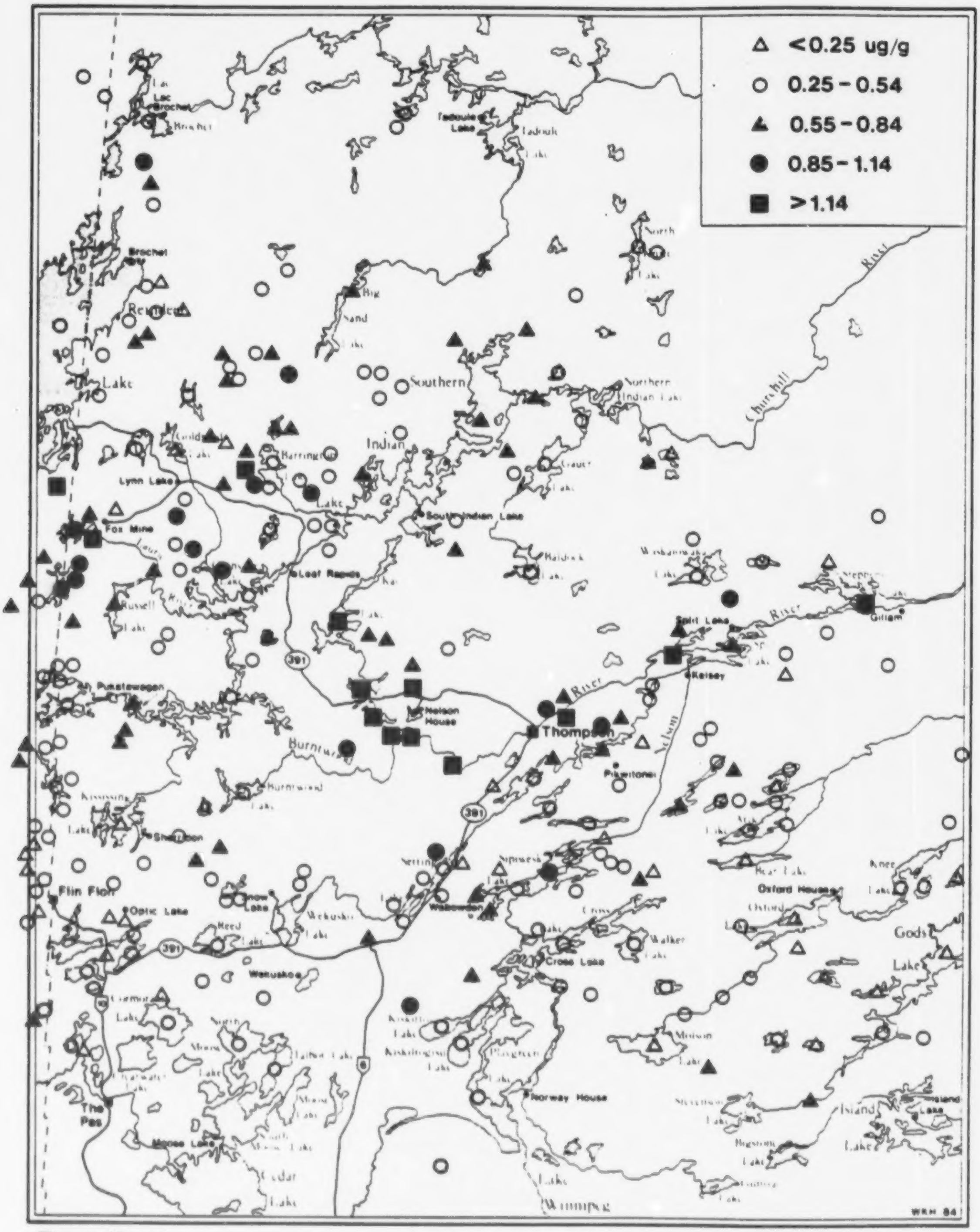


Figure 9: Distribution of lake mean mercury concentrations in commercial samples of northern pike (A) in study region, (B) in northern Saskatchewan.



△ <0.25 ug/g
○ 0.25 - 0.54
▲ 0.55 - 0.84
● 0.85 - 1.14
■ >1.14
Lac Brochet
Brochet
Tadoule Lake
Big Sand Lake
Southern Indian Lake
Northern Indian Lake
Churchill River
Lynn Lake
Fox Mine
South Indian Lake
Leaf Rapids
Reindeer Lake
Gauer Lake
Baldock Lake
Split Lake
Gillam
Kelsey
Nelson House
Thompson
Pukatawagan
Burntwood
Kississing Lake
Sherridon
Flin Flon
Snow Lake
Wekusko Lake
Setting Lake
Sipiwesk Lake
Wabowden
Cross Lake
Walker Lake
Oxford House
Oxford Lake
Knee Lake
Gods Lake
Molson Lake
Wanless
Kiskittogisu Lake
Norway House
Island Lake
The Pas
Moose Lake
Cedar Lake
Lake Winnipeg
Cormorant Lake
Clearwater Lake
Reed Lake
Talbot Lake
Playgreen Lake
Pikwitonei
Optic Lake
391
6
WRH 84

Figure 10: Mean mercury concentration in northern pike, commercial monitoring data.

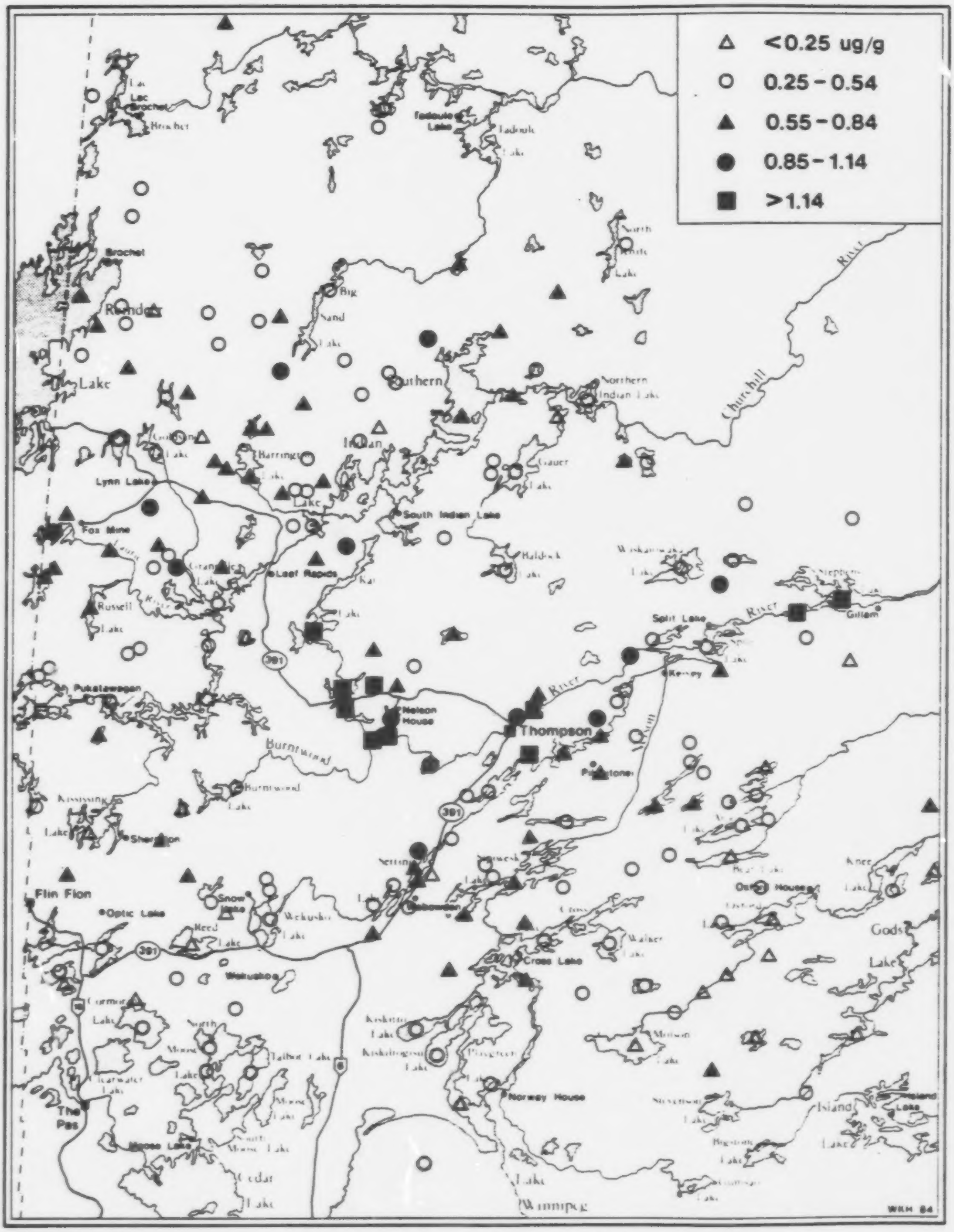

△ <0.25 ug/g
○ 0.25 - 0.54
▲ 0.55 - 0.84
● 0.85 - 1.14
■ >1.14
Lac Brochet
Brochet
Tadoule Lake
North Knife Lake
Churchill River
Reindeer Lake
Big Sand Lake
Southern Indian Lake
Northern Indian Lake
Barrington Lake
Gauer Lake
Lynn Lake
Fox Mine
South Indian Lake
Granville Lake
Leaf Rapids
Russell Lake
Waskaiowaka Lake
Stephens Lake
Split Lake
Gillam
Kelsey
Pukatawagan
Nelson House
Thompson
Burntwood
Burntwood Lake
Paint Lake
Kississing Lake
Sherridon
391
Flin Flon
Optic Lake
Snow Lake
Wekusko Lake
Reed Lake
Wabowden
Cross Lake
Walker Lake
Oxford House
Knee Lake
Gods Lake
Cormorant Lake
North Moose Lake
Clearwater Lake
Talbot Lake
Kiskitto Lake
Kiskittogisu Lake
Playgreen Lake
Norway House
The Pas
Moose Lake
South Moose Lake
Cedar Lake
Lake Winnipeg
Molson Lake
Island Lake
Bigstone Lake
WRH 84

Figure 11: Mean mercury concentration in northern pike, lake survey data.

TABLE 20

Distribution of Lake Mean Mercury Concentrations in Northern Pike in lakes near Flin Flon and East of the Nelson River

CONCENTRATION RANGE (ug/g)	PERCENTAGE OF SURVEYED LAKES		
	WITHIN 150 km OF FLIN FLON	EAST OF NELSON RIVER	REST OF STUDY AREA
Less than 0.24	20.2	26.6	6.4
0.25 to 0.54	65.5	60.6	45.8
0.55 to 0.84	14.3	12.8	29.0
0.85 to 1.14	0	0	11.0
Greater than 1.14	0	0	7.8

Throughout most of the study region, lakes with mean concentrations in the 0.25-0.54 and 0.55-0.84 ug/g ranges are interspersed in apparently random fashion. Occasional anomalously high or low 'inliers' are common, frequently adjacent to lakes with very different concentrations.

2.8 - MAMMALS

Kucera (1983) reported on total mercury contents for mink (*Mustela vison*) and otter (*Lontra canadensis*) tissues in five areas of Manitoba, including Wekusko Lake and the Burntwood River located within the region covered by this report. Data for these two areas are given in Table 21 and a comparison of means in the five areas is given in Table 22. Relatively low fish mercury levels in Wekusko Lake and high levels in the Burntwood River (Part 2.7) are reflected in the data for mink tissues. While no data for otter tissues exist for the Burntwood River, Wekusko Lake again showed low levels in comparison with the Winnipeg River. In addition, Kucera suggested that the slope of the regression of brain mercury on liver plus kidney mercury for mink from the Burntwood River indicated that the exposure to organic mercury there was the highest of the five areas studied, as would be expected from the exceptionally high fish mercury levels observed on the Burntwood River.

TABLE 21

Mercury Levels in Mammals Within Study Region (Kucera, 1983)
(ug/g freeze dried tissue)

AREA	MALE				FEMALE			
	n	LIVER	KIDNEY	BRAIN	n	LIVER	KIDNEY	BRAIN
MINK								
Wekusko Lake	0	2.35±1.74 (0.49-5.12)	2.33±1.55 (0.53-4.81)	0.75±0.89 (0.80-3.13)	9	4.68±3.80 (0.05-10.13)	2.31±2.61 (0.10-8.17)	1.78±0.95 (9.44-2.66)
Burntwood River	15	4.13±2.65 (1.03-10.51)	2.40±1.71 (0.46-7.46)	1.23±0.81 (0.44-3.14)	6	7.31±8.17 (2.01-23.86)	7.02±8.68 (2.03-23.5)	5.28±8.14 (0.69-19.69
OTTER								
Wekusko Lake	10	4.34±2.14 (1.27-8.5)	2.29±6.24 (0.03-4.42)	0.84±0.45 (0.24-1.71)	7	4.14±2.63 (1.42-7.04)	2.77±1.20 (0.97-4.02)	0.71±0.41 (0.04-1.21)
Burntwood River	*	*	*	*	*	*	*	*

* No Data

TABLE 22

Adjusted Mean Mercury Levels in Mink
from Five Areas of Manitoba (Kucera, 1983)
(ug/g freeze dried tissue)

AREA	n	LIVER	KIDNEY	BRAIN
Duck Mountain	43	1.99	1.55	0.58
Wekusko Lake	19	3.57	2.41	1.19
Whiteshell River	27	4.18	3.42	1.29
Burntwood River	21	5.73	4.10	2.41
Winnipeg River	62	9.58	6.93	2.95

2.9 - HUMANS

As part of a national surveillance programme, the Medical Services Branch of Health and Welfare Canada has monitored mercury levels in the hair and blood of Indian residents in the study region since 1971. The results of these tests for the periods 1971-1978 and 1979-1982 are summarized in Table 23 (Health and Welfare Canada, 1979, 1984) for fourteen communities. All data refer to blood mercury level, tests on hair samples having been converted to blood equivalent by standard formulae. The class intervals used to tabulate the data are Medical Service Branch standards for human blood mercury concentrations (Health and Welfare Canada, 1984):

less than 20 ppb - normal acceptable range
20 - 100 ppb - increasing risk
greater than 100 ppb - at risk

The general decrease in the proportion of test results in the higher categories in the 1979-1982 period indicates that the surveillance programme has been effective, rather than that mercury levels in the fish have declined. Nearly half of the "increasing risk" and all of the "at risk" individuals in the 1979-1982 period were in Southern Indian Lake where high fish mercury levels have been well-documented. The six "at risk" individuals there were all identified in 1979; subsequent testing revealed lower levels in the "normal" or "increasing risk" range. By 1982, the maximum blood level observed was in the 50-59 ppb interval.

TABLE 23

Mercury Concentration (ppb) in Residents of the Study Region, 1971-1978 and 1979-1982, Expressed in Blood Equivalent

COMMUNITY	1971 - 1978				1979 - 1982			
	n	<20	20-99	100-199	n	<20	20-99	100-199
1. Brochet	1	1	0	0	25	22	3	0
2. Cross Lake	166	132	34	0	548	505	43	0
3. Fox Lake	-	-	-	-	1	1	0	0
4. God's Lake Narrows	79	41	38	0	38	25	13	0
5. God's River	152	27	125	0	-	-	-	-
6. Granville Lake	36	6	30	0	13	2	11	0
7. Lac Brochet	200	70	129	0	17	8	9	0
8. Nelson House	189	135	52	2	189	126	63	0
9. Oxford House	27	19	8	0	30	29	1	0
10. Pukatawagan	81	58	23	0	30	26	26	0
11. South Indian Lake	-	-	-	-	665	410	249	6
12. Split Lake	51	44	7	0	626	532	94	0
13. Waywayseesajao	-	-	-	-	64	62	2	0
14. York Landing	40	23	17	0	253	221	32	0
	1022	556(54%)	463(45%)	2(0.4%)	2521	1969(78%)	546(22%)	6(0.4%)

2.10 - ATMOSPHERE

No data exist for atmospheric mercury levels within the study area. Indirect evidence discussed elsewhere in the report, however, suggests that elevated levels occur, at least locally, in the vicinity of the smelter at Flin Flon.

PART THREE

EVALUATION OF POTENTIAL SOURCES OF MERCURY

3.1 - INTRODUCTION

The mercury observed in the ecosystems of the study region may originate within the region (endogenous mercury) or may be imported (exogenous mercury) by natural or human agencies. Such a distinction may appear artificial, especially when exogenous mercury is released into the ecosystem at a point and then recirculated. However, the distinction is fundamental to mercury budgetting. Endogenous mercury establishes the "natural" or background levels of mercury wthin a region, although human and/or natural processes may alter the distribution and rate of release. Exogenous mercury elevates the total mercury inventory of the region, producing patterns of abundance which are not related to the natural environment. In many respects, sources of exogenous mercury are more readily identified and often are more succeptible to manipulation, particularly when the mercury has an anthropogenic source.

The distinction between exogenous and endogenous mercury will be followed in this report to clarify the nature of the mercury "problem" and provide a basis for mercury budget studies which may be attempted in the future.

3.2 - EXOGENOUS MERCURY

Mercury originating elsewhere may be introduced into the study region by human or natural agencies. The distinction refers to the mode of transportation at the boundary since mercury carried into the region by a natural agency may be anthropogenic at its source.

3.2.1 - Natural Processes

Mercury may be carried into the study region by glaciers, rivers or atmospheric circulation.

Arguments against significant glacial contribution to the total mercury of the region were presented in Part 2.2. The provenance of drift which entered the region is to the north in areas generally underlain by plutonic granite and granodiorite. Since these rocks appear to have significantly lower mercury content than the bedrock of the study region, it is likely that soils with relatively high mercury content were replaced by drift of reduced mercury, implying perhaps a net loss of mercury due to glaciation.

Only two significant rivers, the Nelson and the Churchill, enter the study region. Elevated mercury levels have been observed in several river systems which feed into Lake Winnipeg and thence into the Nelson River (Derkson, 1978). In the Saskatchewan River (and Cedar Lake), the mercury was attributed to a mercury cell chlor-alkali plant in Saskatoon, with other possible contributors being a pulp and paper mill at Prince Albert and mercury-containing shales (Derkson, 1978). High fish mercury levels have also been observed in the Red and Assiniboine Rivers. Undoubtedly, however, the major contributor of mercury to Lake Winnipeg is the Winnipeg-English-Wabigoon River system where extreme contamination from a mercury cell chlor-alkali plant is well documented (Parks et al., 1984). Derkson (1978) reported that the highest fish mercury levels in Lake Winnipeg occur in the southern basin (which receives water from the Red and Winnipeg Rivers) and around Grand Rapids near the outlet of the Saskatchewan River-Cedar Lake system. Thus contamination from identifiable sources exists in Lake Winnipeg. Some doubt exists, however, as to the extent to which this mercury contributes to the levels observed in fish along the Nelson River downstream. Lower fish mercury levels observed at Long Point (away from the outlet of Cedar Lake) led Derkson (1978) to suggest that the mercury in the north basin of Lake Winnipeg is greatly diluted at the lake outlet. Certainly Lake Winnipeg acts as an efficient settling basin for suspended sediment and little mercury adsorbed onto sediment can enter the Nelson River. Some support for this comes from the low to moderate levels of mercury in fish from Playgreen Lake below the outlet, which Derkson attributed to dilution of locally-generated mercury by Lake Winnipeg water. Further downstream, at Pipestone, Cross, Duck and Sipiwesk Lakes, mercury levels rise rapidly. This pattern led Derkson (1978) to suggest local sources for the contamination, implying that exogenous mercury entering the region via Lake Winnipeg and the Nelson River is of secondary importance only.

The Churchill River enters the study region from Saskatchewan at Loon Lake. Sediments from lakes on the Reindeer-Churchill system in Saskatchewan have elevated mercury concentrations (Part 2.4) and the rivers carry "fairly heavy silt loads during (rising) streamflow periods" (Churchill River Study, 1976, p.25). Although much of the sediment must be lost to lakes in Saskatchewan, some adsorbed mercury may reach the study region. Northern pike in lakes along the Churchill River (from Sokatisewan in Saskatchewan to Highrock in Manitoba) contain 0.4-0.5 ug/g, values which are near the median for the study region and are typical of the entire Churchill River as far as Southern Indian Lake. It appears, then, that whatever mercury is transported along the Churchill River, it is not a fundamental cause of the higher levels further downstream.

The other potential natural source of exogenous mercury is the atmosphere. No measurements of atmospheric mercury concentration are available for the study region but mercury fallout rates in "pristine" environments elsewhere suggest a gradual doubling of atmospheric mercury flux since the beginning of this century due to industrial and other anthropogenic emissions (Kemp et al., 1978). Potential sources of above-background emission outside the region (petroleum refineries in Alberta and Saskatchewan, and until 1975/78 chlor-alkali plants in Dryden and Saskatoon) are probably too distant to have serious impact on the study region. This conclusion is supported by the lake sediment data from the region, which are not elevated; the gradual increase observed in a sediment core from Southern Indian Lake (Part 2.4) falls within global limits. The high concentrations in lake sediments along the Reindeer-Churchill

Rivers appear to be exceptions to this but the much lower values in adjacent areas of Manitoba argue against an atmospheric source. Thus, there is no evidence to suggest that abnormal amounts of mercury are being transported into the study region by the atmosphere.

3.2.2 - Human Processes

Exogenous mercury contained in substances used by man may be released into the environment, adding to the total mercury load within the study region (Table 24). Most of these are of little or no consequence in the present study. No mercury cell chlor-alkali plants have ever been located within the study region, nor have mercury-containing slimicides been used in pulp and paper operations. Agricultural land use is insignificant and limited to the Wabowden Experimental Substation which operated near Wabowden from 1954 to 1966 (Teillet, 1979). The locations of municipal waste disposal sites on federal land are shown in Figure 12. No data are available on the possible mercury content of these and other waste disposal sites but the entire population of the study region is less than 50,000, or about the population of one small city. Of these, approximately 40% are in Thompson which is downstream of most of the study region, 17% are in Flin Flon in the extreme southwest of the region, and the rest are in widely-scattered communities on numerous water systems. Thus municipal waste can be effectively disregarded as a source of mercury loading. From similar reasoning, the contribution from fossil fuel consumption appears insignificant.

TABLE 24

Selected Anthropogenic Sources of Mercury

Industrial Processes

- mercury cell chlor-alkali plants
- pulp and paper manufacture (slimicide)
- gold recovery (amalgamating agent)

Agriculture

- seed dressing (fungicide)
- pesticide

Fossil Fuel Consumption

- thermal electric generators

Miscellaneous

- paint (preservative, pigment)
- electrical equipment
- cosmetics (preservative)
- dental fillings
- domestic waste disposal devices
- paper

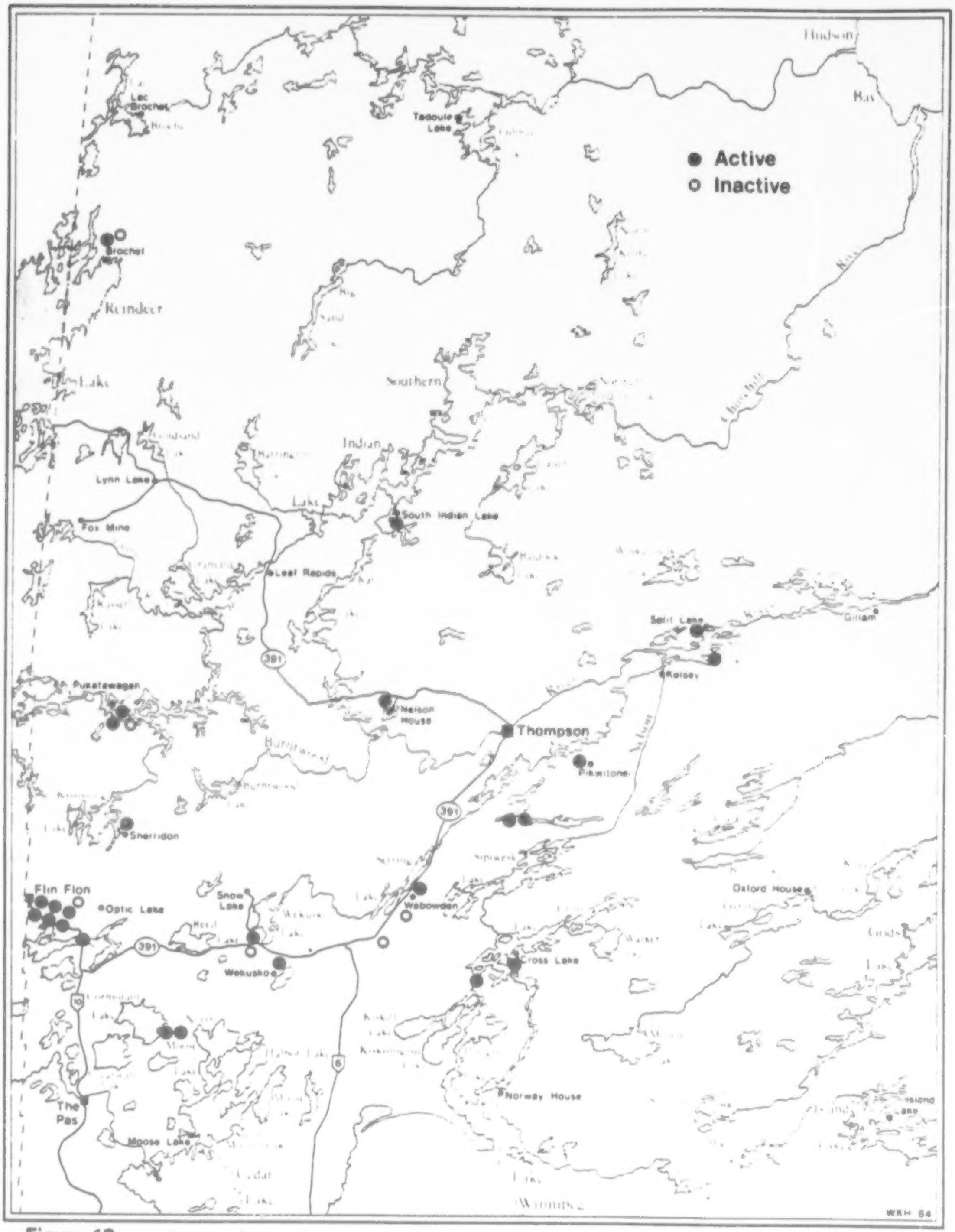

Hudson Bay
● Active
○ Inactive
Lac Brochet
Tadoule Lake
Brochet
Reindeer Lake
Southern Indian Lake
Lynn Lake
Fox Mine
Leaf Rapids
South Indian Lake
Split Lake
Gillam
Kelsey
Nelson House
Thompson
Pikwitonei
Pukatawagan
Sherridon
391
Flin Flon
Optic Lake
Snow Lake
Wabowden
Wekusko
Cross Lake
Oxford House
Norway House
Island Lake
The Pas
Moose Lake
Cedar Lake
Lake Winnipeg
10
6
WKH 84

Figure 12: Active and inactive waste disposal sites on Federal land.

Of the uses in Table 24, the one most likely to contribute exogenous mercury to the study region is gold amalgamation. Several of the gold mines which have been located in the Flin Flon area are known to have used mercury in the amalgamation process and others are assumed to have done so (Table 25, Figure 15).

TABLE 25

Gold Mining Operations in the Study Region Known (K) or Assumed (A) to Have Used Mercury Amalgamation (Sherbin, 1979)

MINE	LOCATION	YEARS OF OPERATION	STATUS
1. Crowduck Ferro	Herb Lake	1932-33, 1962-64 1972-74	K
2. North British	Herb Lake	1933-?	K
3. Century	Elbow Lake	1942-43	K
4. Nor Acme	Snow Lake	1949-58	A
5. Gurney Gold	Optic Lake	1938-39	A
6. Rex Laguna	Wekesko Lake	1918-26, 1936-39	A
7. Pamon	Amisk Lake, Sask.	1937-45	K

The impact of these mines, discussed in some detail in Part 3.3.2, appears to be limited.

In summary, with the possible exception of that from gold recovery operations, mercury imported into the region either by natural or human agencies appears to be insufficient to cause the observed elevated mercury concentrations in fish. Gold amalgamation may have contributed mercury to local water systems in some lakes near Flin Flon, but its total contribution to the mercury inventory of the study region is probably unimportant.

3.3 - SOURCES OF ENDOGENOUS MERCURY

3.3.1 - Bedrock

The relatively high concentrations of mercury in many of the rock types found in the study region were discussed in Part 2.1. Particularly high values occur in association with sulphide mineralization. The massive sulphide ores in northern Manitoba occur in a variety of geologic settings, from volcanics to volcanogenic sedimentary rocks. The association between volcanic rocks and lakes with high fish mercury levels has been noted by Derkson (1978).

Gale et al. (1980) have mapped the potential for occurrences of massive sulphide ore bodies throughout northern Manitoba, including the region covered by this report. Sulphide potential was rated as high (Ratings 1 and 2), medium (3 and 4) or low (5 and 6) according to the dominant lithology. The regions with high sulphide potential are indicated on Figure 13. Given the high mercury contents of the sulphide ores at existing mines, Figure 13 may be taken as a first approximation of areas of potentially elevated mercury levels in the rocks of the study region. It should be noted, however, that even areas with relatively low overall potential (Ratings 5 and 6) may contain small masses of volcanogenic rocks which are undiscovered or overlain by younger sediments but which are nevertheless sufficient to provide a source of locally-elevated mercury.

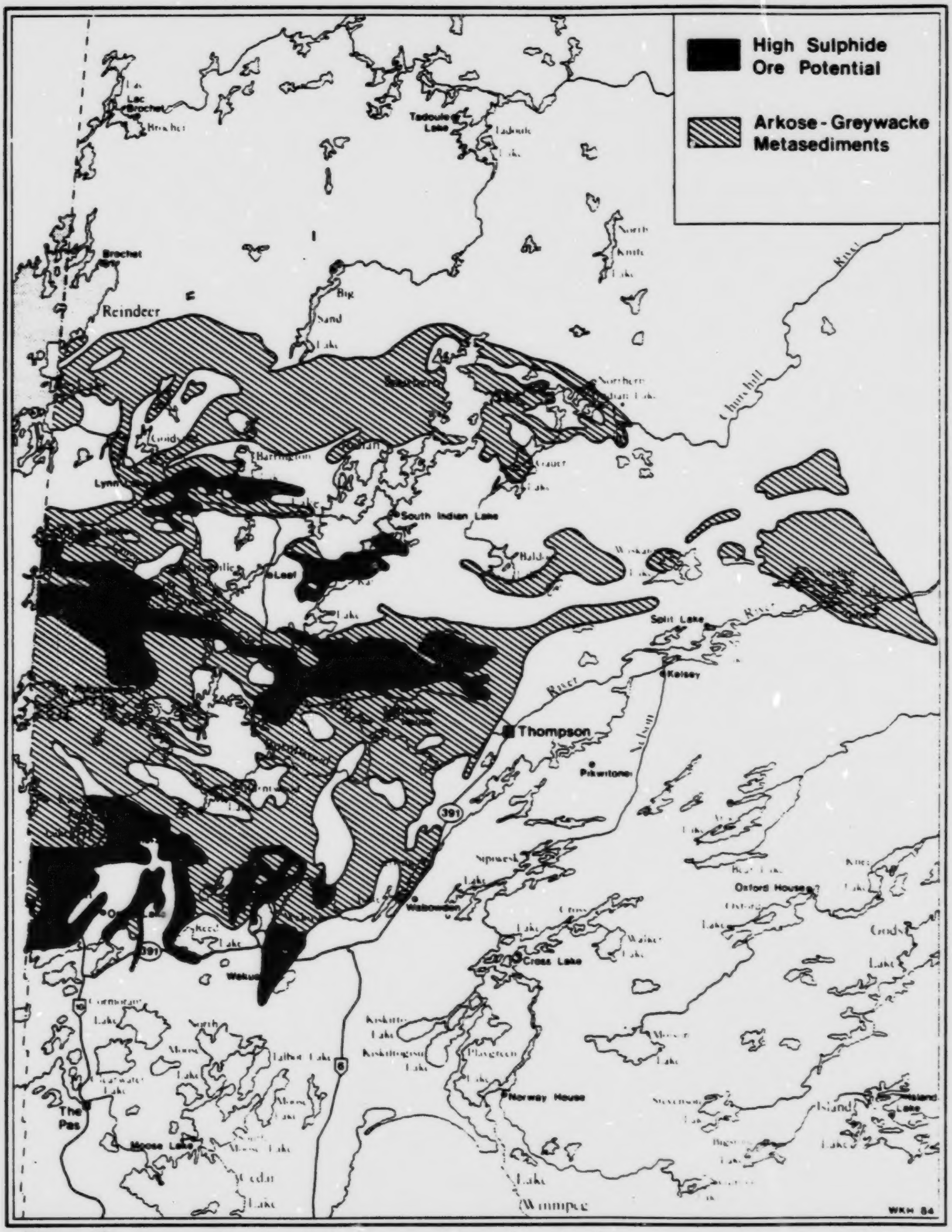

High Sulphide Ore Potential
Arkose - Greywacke Metasediments
Lac Brochet
Brochet
Tadoule Lake
Brochet Bay
North Knife Lake
Big Sand Lake
Reindeer
Churchill
River
Northern Indian Lake
Lynn
Barrington
South Indian Lake
Leaf
Split Lake
Kelsey
Thompson
Pikwitonei
391
Sipiwesk Lake
Wabowden
Oxford House
Cross Lake
Walker Lake
Wekusko
Knee Lake
Gods Lake
Cormorant Lake
Talbot Lake
6
Playgreen Lake
Norway House
Island Lake
The Pas
Moose Lake
Cedar Lake
Lake Winnipeg
WKH 84

Figure 13: Zones of high sulphide ore potential and arkose-greywacke metasediments in study region.

The Superior Province in the southeastern part of the study region is underlain mostly by granodiorite and tonalitic gneisses of plutonic origin with little potential for massive sulphide mineralization and normally having low mercury concent. The mercury concentrations in the fish of this region are generally low (Part 2.7), even in lakes near volcanic rocks which elsewhere are often associated with high concentrations (Derkson, 1978). Similarly, the region east of Southern Indian Lake is dominated by plutonic rocks with relatively low mercury potential and fish in lakes there have generally below-average mercury concentrations.

High sulphide potential exists in a narrow belt extending northeast from the Fox Lake Mine region through Lynn Lake and Barrington Lake to near Southern Indian Lake. The Fox Lake Mine region contains several lakes with elevated fish mercury concentrations and most lakes in this narrow belt are above the average for the study region.

Another, broader, arc of high sulphide potential runs southeast from Fox Lake, south of Granville Lake, through southern Notigi Lake to the vicinity of Muskego Lake north of Thompson. Fish mercury concentrations in this zone are inconsistent, with most of the high values occurring along the Churchill Diversion system. Another zone indicated by Gale et al. (1980) as having high potential occurs at the northern end of the Diversion route astride the Rat River.

The classification devised by Gale et al. (1980) was oriented toward the discovery of potential ore bodies. Most massive sulphide ore bodies tend to be sharply defined in both lateral and vertical extent, and within these strata, the main mercury-bearing minerals are even more restricted (Gale et al., 1980). Consequently, the most mercury-rich bedrock zones can be expected to be very localized. However, minor massive and disseminated sulphide mineralizations are widespread in metagreywacke and meta-arkose, rocks which exhibited elevated mercury levels in the Rice Lake region (Table 2). Particularly favourable zones occur along the contact between the lower Sickle Group and the Burntwood River Metamorphic Suite (Baldwin, 1980). According to Baldwin,

> Disseminated, conformable, stratiform sulphide mineralization occurs at or close to the Sickle Group-Burntwood River Metamorphic Suite contact in Manitoba from the Manitoba - Saskatchewan border eastward to the Mynarski Lakes. Reconnaissance mapping in the Macheewin-Waskaiowaka, Partridge Breast - Northern Indian Lake regions...indicates that Sickle Group-Burntwood River Metamorphic Suite lithologies continue eastward from the Mynarski Lakes to the Thompson Nickel Belt and therefore extends the potential environment for sulphide mineralization to a total strike length of 270 km, and to 450 km if the meta-arkosic rocks in the Long Spruce are correlated with the Sickle Group. (p. 11)

Elevated mercury levels associated with these rocks may be expected to display a similarly widespread distribution (Figure 13). The Rice Lake data also indicate high mercury levels in gabbro and some diorite which occur on a scale too small to be shown on Figure 13.

Comparison of the distribution of lakes with above-average fish mercury concentrations with the bedrock geology indicates that most such lakes occur in regions where elevated rock mercury concentrations may be anticipated. This association is shown in greater detail in Figure 14 for the Lynn Lake-Granville Lake area. The mean mercury concentration of northern pike in lakes dominantly underlain by rocks of the Sickle and Wasekwan Groups is 0.93 ug/g (range - 0.53-1.47 ug/g) and all lakes exceeded the standard for domestic sale (0.50 ug/g). In the other lithologic regions shown on Figure 14, the mean concentration is 0.53 ug/g (range - 0.22-0.94 ug/g) with 56% less than 0.50 ug/g.

Bedrock lithology, then, appears to be the basic determinant of general fish mercury concentrations within the study region. Exceptions are common, however, notably in the Flin Flon region. The Flin Flon-Snow Lake-Wekusko Lake Belt of volcanics and volcanogenic sediments contains many sulphide ore bodies, most with significant quantities of sphalerite, and since 1930, much of the mining activity in the study region has been located in this zone. The apparently elevated mercury content of these ores (Part 2.1) is not reflected in the fish of the area. As was noted in Part 2.7, fish mercury concentrations in the Flin Flon area are below the study region average and none of the seriously contaminated lakes occur in this zone. Thus, lithology by itself is an imperfect guideline to fish mercury concentrations in nearby lakes. Other factors must intervene to facilitate or inhibit the disseminiation of mercury into the biosphere.

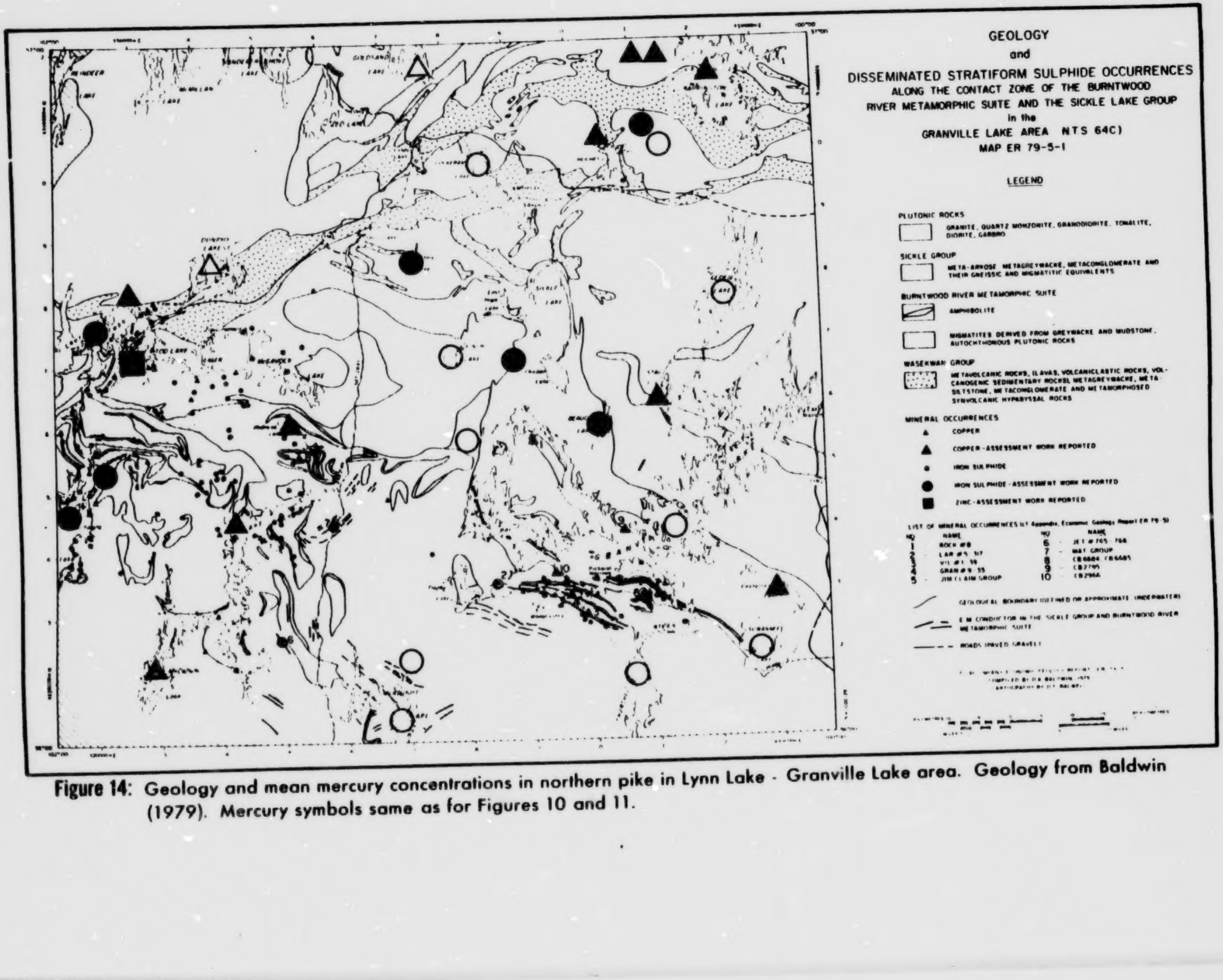

GEOLOGY
and
DISSEMINATED STRATIFORM SULPHIDE OCCURRENCES
ALONG THE CONTACT ZONE OF THE BURNTWOOD
RIVER METAMORPHIC SUITE AND THE SICKLE LAKE GROUP
in the
GRANVILLE LAKE AREA NTS 64C)
MAP ER 79-5-1
LEGEND
PLUTONIC ROCKS
GRANITE, QUARTZ MONZONITE, GRANODIORITE, TONALITE, DIORITE, GABBRO
SICKLE GROUP
META-ARKOSE METAGREYWACKE, METACONGLOMERATE AND THEIR GNEISSIC AND MIGMATITIC EQUIVALENTS
BURNTWOOD RIVER METAMORPHIC SUITE
AMPHIBOLITE
MIGMATITES DERIVED FROM GREYWACKE AND MUDSTONE, AUTOCHTHONOUS PLUTONIC ROCKS
WASEKWAN GROUP
METAVOLCANIC ROCKS, (LAVAS, VOLCANICLASTIC ROCKS, VOLCANOGENIC SEDIMENTARY ROCKS), METAGREYWACKE, METASILTSTONE, METACONGLOMERATE AND METAMORPHOSED SYNVOLCANIC HYPABYSSAL ROCKS
MINERAL OCCURRENCES
▲ COPPER
▲ COPPER - ASSESSMENT WORK REPORTED
● IRON SULPHIDE
● IRON SULPHIDE - ASSESSMENT WORK REPORTED
■ ZINC - ASSESSMENT WORK REPORTED
LIST OF MINERAL OCCURRENCES (cf Appendix, Economic Geology Report ER 79-5)
NO NAME
1 - ROCK #8
2 - LAR #5 317
3 - VIL #1 39
4 - GRAN #9 55
5 - JIM CLAIM GROUP
NO NAME
6 - JET #765 766
7 - MAT GROUP
8 - CB6684 CB6685
9 - CB2795
10 - CB2984
GEOLOGICAL BOUNDARY (DEFINED OR APPROXIMATE, INDEPRMATED)
E M CONDUCTOR IN THE SICKLE GROUP AND BURNTWOOD RIVER METAMORPHIC SUITE
ROADS (PAVED GRAVEL)
COMPILED BY D.A. BALDWIN 1979
GIRDLANDS LAKE
REINDEER LAKE
SICKLE LAKE
10

Figure 14: Geology and mean mercury concentrations in northern pike in Lynn Lake - Granville Lake area. Geology from Baldwin (1979). Mercury symbols same as for Figures 10 and 11.

3.3.2 - Drainage from Mine Tailings

Tailings from active and inactive mines occur in many parts of the study region (Figure 15). The ore from these mines may contain significant amounts of mercury and, if the ore has been concentrated, these levels may be further elevated. In the case of gold mines, exogenous mercury used in the amalgamation process may also be present. The permeability of these tailings or concentrates is several orders of magnitude higher than the parent bedrock and the leachate is frequently more acidic than normally circulating soil and groundwater. According to Jonasson and Boyle (1972)

> acidity is known to be a critical factor... the highest concentrations of mercury in ground waters are associated with highest acidity... Mercury ions (-ic and -ous) or their hydrolyzed colloidal aggregates adsorb strongly on surfaces of particulate materials between pH 4 and 7 (p. 26)

Thus there are sound reasons for suspecting that mine tailings drainage contributes to elevated mercury levels in adjacent water bodies and ecosystems. Some indication of the potential impact of these tailings dumps is provided by impact studies conducted in recent years at several abandoned mine sites in northern Manitoba.

Beck (1983) studied the impact of mine drainage on lakes near the abandoned Sherridan copper-zinc mine which operated from 1928 to 1951. Pyrite was the most abundant sulphide, followed in decreasing order by pyrrhotite, chalcopyrite and sphalerite. The tailings disposal area occupies approximately 44 hectares adjacent

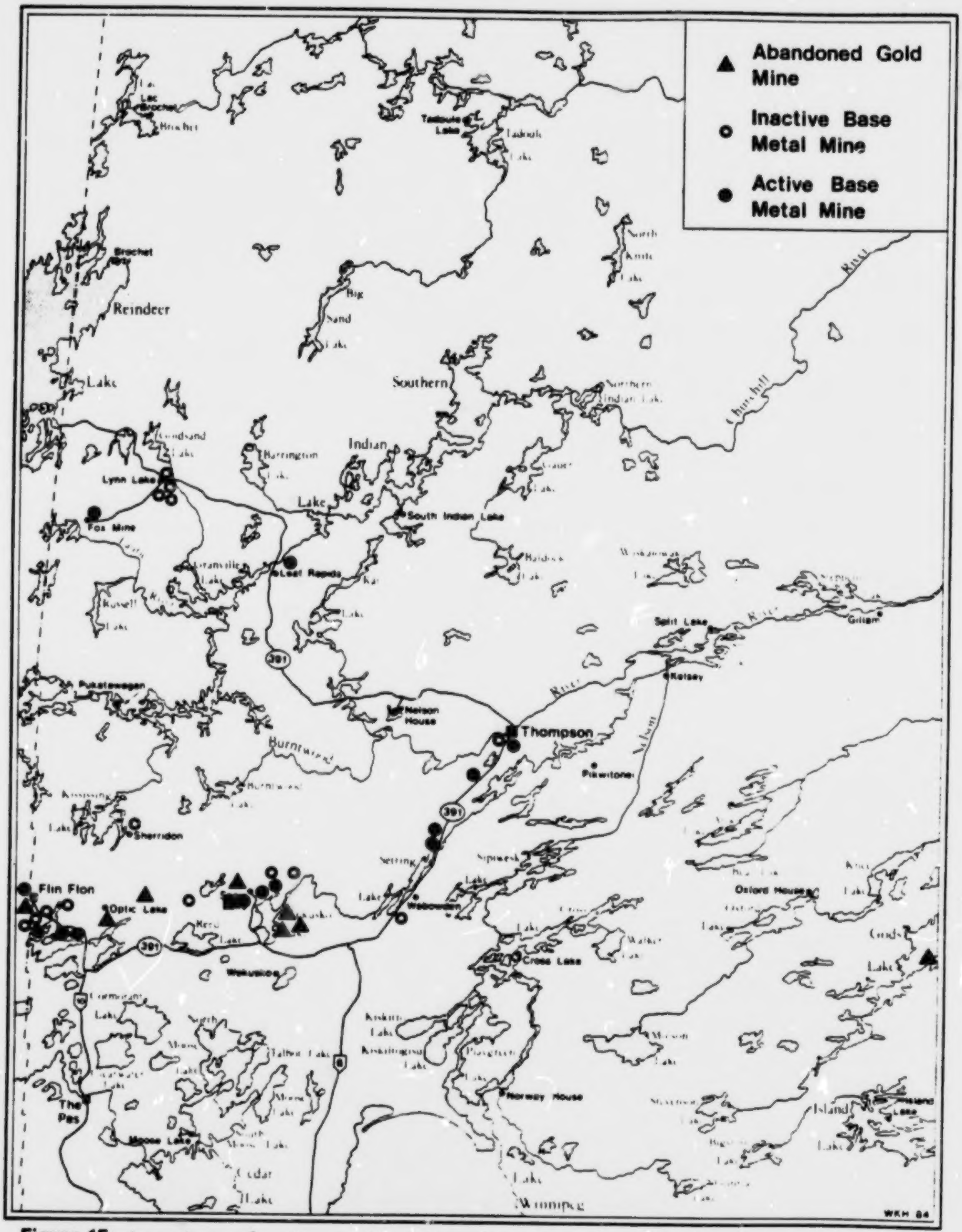

▲ Abandoned Gold Mine
○ Inactive Base Metal Mine
● Active Base Metal Mine
Lac Brochet
Brochet
Tadoule Lake
Brochet Bay
Reindeer Lake
Big Sand Lake
North Knife Lake
Southern Indian Lake
Northern Indian Lake
Churchill River
Barrington Lake
Lynn Lake
Fox Mine
South Indian Lake
Leaf Rapids
Granville Lake
Rusell Lake
Split Lake
Gillam
Kelsey
391
Pukatawagan
Nelson House
Thompson
Burntwood
Pikwitonei
Kississing Lake
Sherridon
Setting Lake
Sipiwesk Lake
Flin Flon
Optic Lake
Kero Lake
Wabowden
Cross Lake
Walker Lake
Oxford House
Gods Lake
Wekusko
Cormorant Lake
Talbot Lake
Kiskitto Lake
Kiskittogisu Lake
Playgreen Lake
Norway House
The Pas
Moose Lake
Cedar Lake
Island Lake
Lake Winnipeg
WKH 84

Figure 15: Locations of active and inactive mines in study region (various sources).

to Camp Lake. Water, lake sediment, and fish samples were taken from four localities - Camp Lake adjacent to the main drainage path from the tailings, "Block A" on Cold Lake at the outfall from Camp Lake, "Block B" further downstream on Cold Lake, and Sherlett Lake, a control site upstream from Camp Lake assumed to be unaffected by tailings drainage. The tailings have high mercury contents (Table 26), and this is reflected in the elevated mercury in the Camp Lake sediments (the highest known mercury content in any lake sediments in the study region - see Table 15). While the sediments of other lakes in Table 26 contain much less mercury than Camp Lake, they are nonetheless greatly elevated in comparison with lakes elsewhere in the study region. The other striking feature of Camp Lake is its very low pH (Table 27), indicative of the very acid nature of the drainage from the sulphide tailings. This high acidity undoubtedly plays an important role in mobilizing the mercury in these tailings (Jonasson and Boyle, 1972). The acidity of Camp Lake is transferred to Cold Lake Block A where a significant correlation was observed between pH and distance from the outfall from Camp Lake. No fish were found in Camp Lake (due to the high acidity and metal loading) but the mercury contents of fish from Cold Lake downstream do not suggest serious impact. Northern pike averaged 0.22 ug/g, the same as in the control lake (Sherlett) and no pike exceeded the domestic commercial standard for mercury content. The data from Table 28 suggest that at Sherridan, significant bioaccumulation of mercury from tailings drainage does not occur beyond the receiving lake, i.e. beyond about one kilometer.

TABLE 26

Mercury Concentrations (ppb) in Lake Sediments
Near Sherridan Mines (Beck, 1983)

LAKE	n	MEAN (ppb)	STANDARD DEVIATION	RANGE
Sherlett Lake	15	320	130	<30-660
Camp Lake	15	1140	690	420-2420
Cold Lake A	14	370	400	50-1460
Cold Lake B	13	460	270	30-790
Tailings	4	2810	2200	1190-5960

TABLE 27

pH of Lake Waters near Sherridan Mine (Beck, 1983)

LAKE	n	LOGARITHMIC MEAN	STANDARD DEVIATION	RANGE
Sherlett Lake	15	7.45	0.14	7.05-7.60
Camp Lake	15	4.21	0.74	3.20-6.30
Cold Lake A	15	6.20	1.02	4.10-6.55
Cold Lake B	14	6.97	0.10	6.75-7.10

TABLE 28

Mercury Concentrations in Fish From Lakes Near Sherridan Mine
(Beck, 1983)

LAKE	SPECIES	n	MEAN ug/g	STANDARD DEVIATION	RANGE
Sherlett Lake	N. Pike	20	0.22	0.17	<0.02-0.46
	Walleye	20	0.64	0.29	0.27-1.17
	Common Sucker	20	0.10	0.07	0.05-0.27
Cold Lake	N. Pike	7	0.22	0.08	0.13-0.35
	Walleye	9	0.14	0.04	0.05-0.19
	Common Sucker	20	0.04	0.02	0.02-0.08

Beck (1984) examined the impact on Snow and Herblett Lakes of mine drainage from the Nor-Acme gold mine which operated from 1949 to 1958. Leachates from stockpiled concentrates and tailings drain toward Snow Lake and Herblett Lake respectively. The mercury content of the concentrate stockpile is relatively small (Table 29) and the mean mercury content of Snow Lake sediments (Table 30) is similar to values reported in lakes elsewhere in the study region. Thus, while the leachate from the concentrate stockpile is very acidic (Table 31), it appears to mobilize relatively little mercury. The small amount of mercury contained in the tailings (Table 29) is reflected in the very low mercury content of the sediments in Herblett Lake (Table 30) and Beck (1984) concluded that there was no indication that tailings had transported significant quantities of mercury to Herblett Lake. Mean mercury levels in fish were within federal limits, with only one fish (a northern pike from Herblett Lake) exceeding 0.50 uq/g.

The impact of gold mining on Wekusko Lake was studied by Brown et al. (1980). Several mines have been located in the Wekusko Lake area, most notably the Laguna Mine which operated intermittently during the years 1918-25 and 1936-39. The ore contained small amounts of galena, sphalerite and pyrite sulphides (Davies et al., 1962) and according to Brown et al., (1980), mercury was used to recover the gold. The principal receiving body for drainage from Laguna mine waste is Crowduck Bay, an arm of Wekusko Lake. From Table 33, the mercury content in Crowduck Bay sediments is highly

TABLE 29

Mercury Concentrations (ppb, dry weight) in Nor-Acme Mine Concentrate and Tailings (Beck, 1984)

Concentrate stockpile surface	- 130
Stream bed at edge of stockpile	- 100
Stream bed c 200 m from stockpile	- 80
Stream bed c 300 m from stockpile	- 110
Tailings surface, Sample 1	- 30
Tailings surface, Sample 2	- <20
Tailings surface, Sample 3	- <20

TABLE 30

Mercury Concentrations (ppb) in Lake Sediments near Nor-Acme Mine (Beck, 1984)

LAKE	n	MEAN (ppb)	STANDARD DEVIATION	RANGE
Snow Lake A	10	78	53	20-160
Snow Lake B	10	68	44	<20-120
Snow Lake C	10	110	46	30-170
Herblett Lake D	10	<20	0	-
Herblett Lake E	10	30	32	<20-120
Herblett Lake F	10	50	72	20-190

TABLE 31

pH of Lake Waters near Nor-Acme Mine (Beck, 1984)

LAKE	n	MEAN	RANGE
Snow Lake A	10	7.29	6.75-7.55
Snow Lake B	10	7.68	7.00-7.90
Snow Lake C	10	7.88	7.80-8.00
Herblett Lake D	10	7.67	7.55-7.80
Herblett Lake E	10	7.93	7.80-8.40
Herblett Lake F	10	7.27	7.15-7.45
Concentrate Drainage	1	3.25	-
Tailings Drainage	1	8.15	-

elevated in comparison with the other areas of Wekusko Lake and it is apparent that a significant quantity of mercury has found its way to the lake. The mercury contents of the sediments in the rest of the lake are normal for the study region, however, and the impact of mine drainage is limited to the immediate vicinity of Crowduck Bay. Fish from Wekusko Lake have relatively low mercury concentrations (Table 34) and provide no evidence of important bioaccumulation; only one fish out of 283 (a sauger) exceeded 0.5 ug/g. Brown et al. (1980) also noted slightly elevated mercury content in sediments in Anderson Lake which they attributed to discharge and leaching from a nearby base metal mine (Anderson Lake Mine, HBMS) which began production in 1971 and/or natural ore bodies. However, the low mercury in Anderson and Stall Creeks (Table 33) suggested that downstream areas were unaffected. There was no significant difference between mercury levels in the creeks and Anderson Bay (the area of Wekusko Lake receiving drainage from Anderson Lake), nor between Anderson Bay and other areas of Wekusko Lake (apart from Crowduck Bay discussed above).

God's Lake is not within the study region but a detailed study of the potential pollution from an abandoned gold mine there by Beck (1982) provides further data on the effects of mining activity on the mercury levels in nearby water bodies and aquatic ecosystems. Gold was mined and milled on Elk Island on the south shore of God's Lake from 1935 to 1943. Mercury was used to extract the gold (Beck, 1982). Tailings were disposed over an area measuring approximately

TABLE 32

Mercury Concentrations (ug/g) in Fish from Lakes Near Nor-Acme Mine (Beck, 1984)

LAKE	SPECIES	n	MEAN	σ	RANGE
Snow Lake	Northern Pike	14	0.31	0.10	0.13-0.45
	Walleye	20	0.21	0.09	0.13-0.48
	Whitefish	20	0.12	0.06	0.04-0.25
	Common Sucker	17	0.04	0.02	<0.02-0.09
Herblett Lake	Northern Pike	20	0.36	0.11	0.19-0.65
	Cisco	20	0.12	0.07	0.04-0.33
	Common Sucker	20	0.07	0.03	0.03-0.13

TABLE 33

Mercury Concentration in Lake Sediments near Laguna Mine (Brown et al., 1980)

LAKE	MEAN CONCENTRATION (ppb)
Wekusko Lake, Anderson Bay	60
Berry Bay	80
Crowduck Bay	510
Goose Bay	50
Herb Bay	70
Anderson Lake	110
Anderson and Stall Creeks	70

TABLE·34

Mercury Concentration (ug/g) in Fish from Wekusko Lake

SPECIES	n	MEAN (ug/g)	STANDARD DEVIATION
Northern pike	30	0.17	0.01
Walleye	37	0.17	0.01
Whitefish	31	0.05	<0.01
Sauger	39	0.31	0.02
Cisco	40	0.09	<0.01
Yellow perch	35	0.08	0.01
Common sucker	40	0.07	<0.01
Longnose sucker	30	0.05	<0.01

1000 km by 150 m. Sampling was conducted by Beck in three blocks-a control Block (A) at the influent area of God's Lake (approximately 50 km uplake from the mine site), a Block (C) adjacent to the mine and tailings disposal site (potentially the most seriously affected by mine activity), and a Block (B) at the lake outlet approximately 17 km across the lake from the mine site. The mercury content of water in God's Lake (Table 35) appears anomalous, assuming that the data as reported by Beck are reliable. In contrast to most water quality data from northern Manitoba (Part 2.5), including those from other inactive mine sites, God's Lake water contains elevated mercury concentrations, with the highest level at the control site (A) and the lowest adjacent to the tailings disposal area. The highest content in Block A (1.2 ug/L) is extremely high and probably unreliable. If it is removed from the Block A samples, the recalculated mean is reduced to 0.03 ug/L, a value more in keeping with the other blocks and with normal values reported elsewhere in northern Manitoba. This seems to be the most reasonable interpretation of the data in Table 35. Samples taken on the tailings pile (Table 37) exhibit evidence of leaching of mercury from the surface layer of the tailings (compare the concentration of 2.58 ug/g in a sample from a depth of 20-30 cm near the top of the pile with values of 0.06, 0.26 and 0.64 ug/g from surface samples). The mercury contents of the lake sediments in Blocks A and B (Table 36) are rather low by northern Manitoba standards but clearly elevated (by an order of magnitude) in Block C which receives tailings drainage. The mean for Block C is deceptive however, since

TABLE 35

Mercury Concentration (ug/L) and pH of God's Lake Water (Beck, 1982)

SAMPLE BLOCK	MEAN	STANDARD DEVIATION	RANGE	MEAN pH
A	0.105	0.303	<0.02-1.2	7.9
B	0.039	0.045	<0.02-0.18	8.0
C	<0.02	-	-	7.9

TABLE 36

Mercury Concentrations (ppb) in Lake Sediment in God's Lake (Beck, 1982)

SAMPLE BLOCK	n	MEAN (ppb)	S.D.	RANGE
A	15	30	12	<20-60
B	15	<20	-	-
C	9	300	510	<20-1570

TABLE 37

Mercury Concentrations (ppb, dry weight) in Sediments in God's Lake Tailings Disposal Area (Beck, 1982)

LOCATION	SAMPLE DEPTH	CONCENTRATION (ppb)
Tailings Centre	20-30 cm	2580
Tailings Edge 1	Surface	640
Tailings Edge 2	Surface	260
Tailings Edge 3	Surface	60
Stream bed, c. 100m from tailings	Surface	960

mercury concentrations ranged from an extremely high value of 1570 ppb to less than 20 ppb and Beck (1982) found a virtually perfect correlation ($r^2 = 0.99$) between mercury content and distance from the mouth of the stream draining the tailings. Block C lies in an embayment protected by a chain of islands. These islands, lake floor topography, and localized currents appear to inhibit normal dispersal of materials carried from the tailings into the main body of God's Lake (Beck, 1982). Consequently, Beck concluded that the total area contaminated with mercury and other metals from the tailings disposal area is limited to approximately 0.05 km^2 of lake surface. Fish from all sample blocks exhibited low to moderate levels of mercury, with average concentrations well below 0.50 ug/g. Only 6 of 230 fish exceeded 0.50 ug/g (one pike from each block and three walleye from Block B). Beck concluded that the data provided no indication that mercury from the tailings had affected fish in God's Lake.

These studies suggest that, while mercury leached from mine tailings does accumulate in the bottom sediments of nearby lakes, the effects are generally confined to the immediate vicinity of the tailings drainage inflow. The mercury contents of fish in all studies were well within federal domestic standards and exhibit little or no evidence of serious bioaccumulation. Some caution must be exercised, however, in extrapolating these conclusions to active mine sites. The mines were all relatively small and had been inactive for several decades during which time much of the mercury may have been mobilized in the bottom sediments, passed out of the system in the water column, or gradually dispersed through several generations of fish. Furthermore, with the important exception of the Sherridan mine, all tailings were from gold bearing ores in which the principal mercury-bearing sulphide, sphalerite, occurred in relatively small amounts. It is clear, however, that inactive gold mine tailings are not a cause of the elevated mercury levels observed in fish in the study region.

Information regarding the mercury impact of base metal mine tailings is somewhat more scanty. The greatest impact in the studies referred to above occurred in Camp Lake adjacent to the Sherridan copper-zinc mine. The mercury content of the tailings was high and that of the Camp Lake sediments exceptionally so. Even thirty years after active mining ceased, the acidity of the lake remained high. Some of this impact was transferred to Cold Lake downstream but again the elevated mercury levels and depressed pH were concentrated near the outfall of Camp Lake. The mercury levels in fish in Cold Lake were apparently unaffected. Studies of the impact of base metal mines elsewhere in the region generall make little reference to mercury. A value of 6.0 ug/L (pH 4.0) is reported by van Loon and Beamish (1977) for mercury in lake water in the grossly contaminated Ross Lake which receives both effluent drainage and atmospheric fallout from the Flin Flon smelter complex. This is two to three orders of magnitude greater than the range normally observed for natural waters. Some measurements of water mercury contentrations near the Fox Lake copper-zinc mine are given in Table 39 (Hughes, 1977). The waters draining the tailings have elevated mercury levels and are very acidic but there is little indication that these characteristics are passed on to lakes downstream. Sediment mercury contents for several of these lakes abstracted from the National Geochemical Reconnaissance Map 64C (Table 40) show values which are somewhat elevated in comparison with the control lake (Dunphy Lake) but which are near the mean for the region. Blunt (1977) reported mercury concentrations of 0.200 ug/L for nine water sample sites on Eldon Lake, Lynn River, Keewatin River and Cockeram Lake, water bodies which are liable to be affected by tailings effluent from the A and El mines near Lynn Lake. Although several sites showed moderately elevated levels of copper, zinc and nickel, these were dissipated relatively rapidly downstream. Mercury levels abstracted from the Geochemical Reconnaissance Map 64C are normal for the study region (Table 40) and mercury concentrations in northern pike in Cockeram Lake averaged only 0.26 ug/g.

These relatively limited data suggest that, whatever their other effects on metal loading and acidity in nearby lakes, effluents from base metal mine tailings are not significant contributors to the elevated mercury levels observed in the aquatic ecosystems of the study region. The heavily contaminated lakes near Flin Flon which receive atmospheric and terrestrial inputs are undoubted exceptions.

TABLE 38

Mercury Concentrations (ug/g) in Fish, God's Lake (Beck, 1982)

SAMPLE BLOCK	SPECIES	n	MEAN	σ	RANGE
A	Northern pike	20	0.30	0.10	0.13-0.51
	Whitefish	7	0.06	0.03	0.03-0.10
	Walleye	20	0.18	0.07	0.10-0.32
	Common Sucker	20	0.06	0.02	0.03-0.10
B	Northern pike	20	0.26	0.22	0.08-1.13
	Whitefish	20	0.10	0.05	0.03-0.20
	Walleye	8	0.48	0.13	0.33-0.70
	Longnose Sucker	20	0.09	0.05	0.03-0.21
C	Northern pike	20	0.29	0.09	0.11-0.52
	Whitefish	20	0.11	0.06	0.03-0.26
	Walleye	15	0.20	-	0.11-0.27
	Common Sucker	20	0.17	0.10	0.04-0.35
	Longnose Sucker	20	0.10	0.07	0.05-0.26

TABLE 39

Mercury Concentrations and pH of Waters near Fox Lake Mine, 1976 (Hughes, 1977)

SAMPLE LOCATION	MERCURY CONCENTRATION (ug/L)	pH	COMMENT
Tailings effluent	1.000	-	
Tailings effluent	0.200	-	
Oz Lake	0.800	3.5	Receives tailings effluent
Jam Lake	<0.200	3.6	Receives tailings effluent
Fox Lake	<0.200	3.8	Downstream of Jam Lake
Snake Lake	0.200,<0.200	6.1,6.8	Downstream of Fox Lake
Dunphy Lake	<0.200	7.0	Unaffected by tailings

TABLE 40

Mercury Concentrations in Lake Sediments Near Fox Lake and Lynn Lake Mines (Geological Survey of Canada, 1984)

LAKE	CONCENTRATION (ppb)	n	RANGE	pH*
Oz Lake	60	1	-	3.5
Fox Lake	70	1	-	3.8
Snake Lake	70	1	-	6.1, 6.8
Dunphy Lake	38	4	30-40	7.0
Eldon Lake	65	2	60-70	
Cocheram Lake	50	5	30-70	

* from Hughes (1977) and Blunt (1977)

3.3.3 - Smelters

Base metal smelters are estimated to contribute about 63% of the total mercury emissions to the atmosphere, from all sources, in Western Canada (Environment Canada, 1983). Ores from base metal mines in the study region are processed at smelters operated by the Hudson Bay Mining and Smelting Company at Flin Flon and the INCO Metals Co. at Thompson (Figure 16) which began operations in 1930 and 1961 respectively. The stack emissions from these smelters are not monitored for mercury and thus no data exist to directly evaluate the mercury loading of the atmosphere from these point sources. However, the ores processed at Flin Flon are known to contain relatively large quantities of mercury, particularly in the concentrates (Part 2.1) and there is abundant evidence from the environment in the immediate vicinity of the smelter to indicate that fallout from the stack is a major source of mercury and other contaminants. Nickel ores of the type processed at Thompson contain little or no sphalerite and the dominant sulphides present are normally much lower in mercury content (Gale et al., 1980; Environment Canada 1983).

The elevated mercury levels in some shrub components, and especially in the soil surface organic layer, around the HBMS smelter at Flin Flon have been discussed in Part 2.3 above. Shewchuck (1983) measured the mercury concentration in snow in northern Saskatchewan and in more detail near Flin Flon (Figure 17A, B). Concentrations of less than 0.1 ug/L (and more commonly less

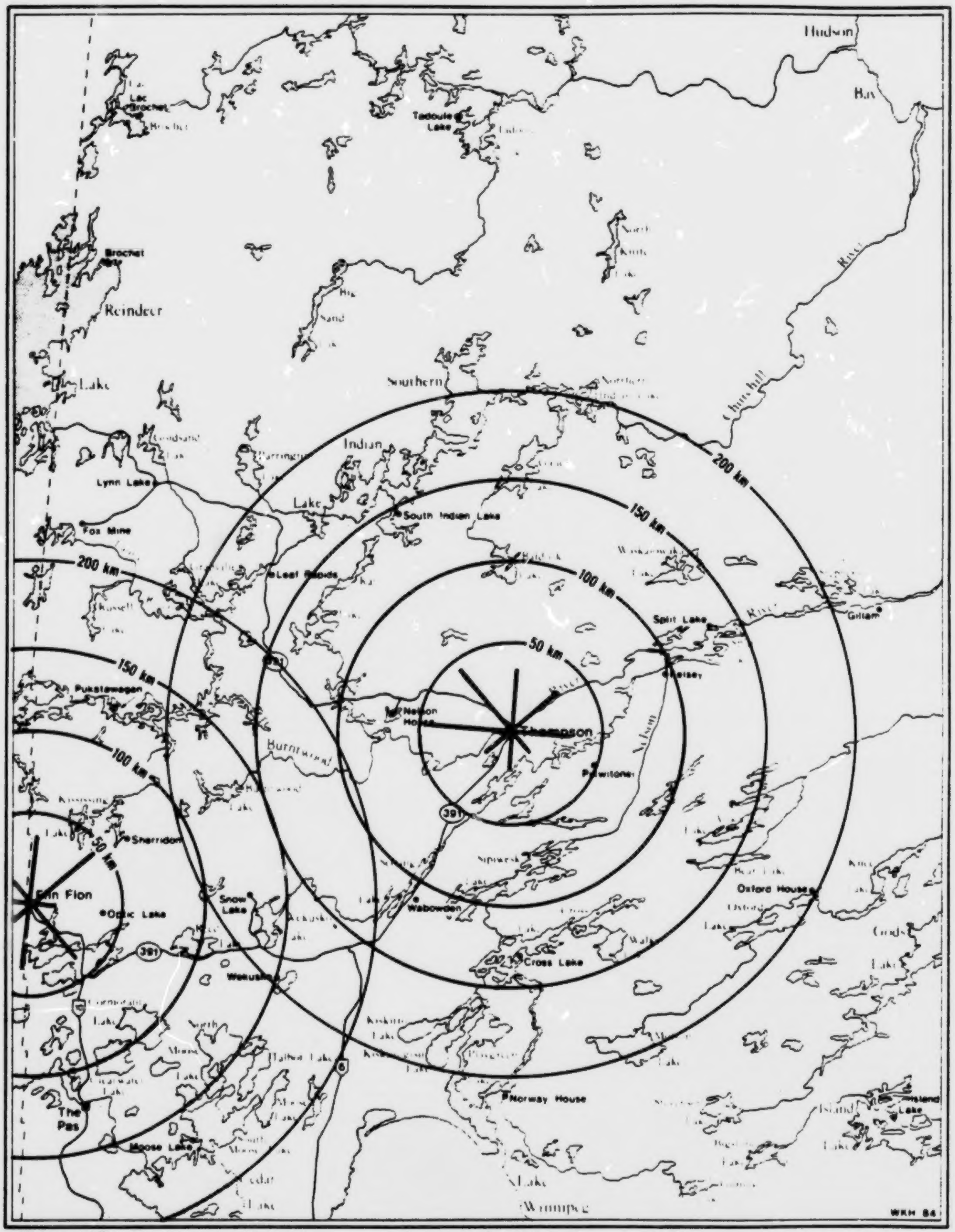

Hudson
Bay
Lac Brochet
Tadoule Lake
Brochet
Reindeer
Lake
Big Sand Lake
North Knife Lake
Southern
Indian
Lake
Churchill
River
Lynn Lake
Fox Mine
Leaf Rapids
South Indian Lake
200 km
150 km
100 km
50 km
Split Lake
Gillam
Kelsey
Nelson House
Thompson
Burntwood
Pikwitonei
Nelson
391
Pukatawagan
Sherridon
Flin Flon
Optic Lake
Snow Lake
Wabowden
Oxford House
Cross Lake
Wekusko
Gods Lake
6
The Pas
Moose Lake
Norway House
Island Lake
Lake
Winnipeg
WRH 84

Figure 16: Locations of smelters, wind roses for Flin Flon and Thompson.

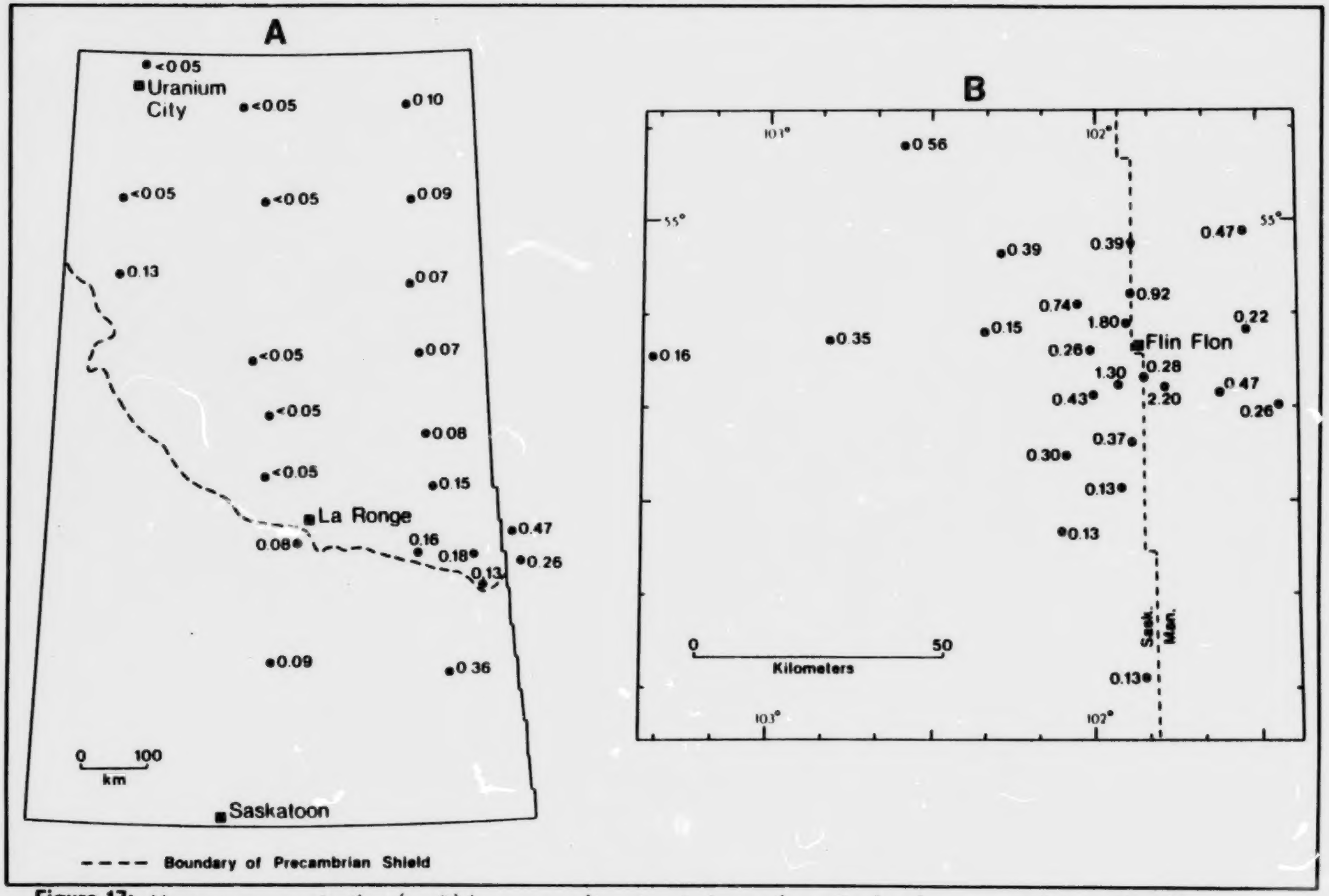

A
<0 05
Uranium City
<0 05
0 10
<0 05
<0 05
0 09
0.13
0 07
<0 05
0 07
<0 05
0 08
<0 05
0 15
La Ronge
0.08
0.16
0.18
0.13
0.47
0.26
0.09
0 36
0 100 km
Saskatoon
Boundary of Precambrian Shield
B
101°
102°
0 56
55°
0 39
0.39
0.47
0.92
0.74
1.80
0.22
0 35
0.15
0.26
Flin Flon
0 16
1.30
0.28
0.47
0.43
2.20
0.26
0.37
0.30
0.13
0.13
0
50
Kilometers
Sask.
Man.
0.13
103°
102°

Figure 17: Mercury concentration (ug/L) in snowmelt water A. in northern Saskatchewan, and B. in Flin Flon region (redrawn from Shewchuk, 1983).

than 0.05 ug/L) were observed throughout most of Sasktchewan and this may be taken as a "background" level. Near Flin Flon, however, several values in excess of 1.0 ug/L were recorded. Greatly elevated values of other metals around Flin Flon have been reported in snow and rainfall (Franzin et al., 1979; Shewchuk, 1983; Hogan and Wotton, 1980), in lake water (van Loon and Beamish, 1977), in fruiting shrubs (Shaw, 1981), in tree and shrub foliage (Hogan and Wotton, 1980, 1984) and in the surface organic layer (Hogan and Wotton, 1980, 1984). In virtually all cases, the correlation coefficients between metal content and distance from the smelter were extremely high (generally greater than -0.90 and in some instances approaching -1.0). Hogan and Wotton (1984) observed that metal levels in the soil surface organic layer were as much as ten times greater than those in lower mineral horizons near the smelter, but that the difference became less pronounced or negligible at more distant sites.

Mercury was measured in only a few of these studies. However, Hogan and Wotton (1984) found high correlations among copper, zinc and lead in the surface organic layer. The data from McEachern and Phillips (1983) indicate correlations of r = 0.70 and r = 0.84 between mercury and lead in the surface organic layer, and in the snow data reported by Shewchuk (1983) mercury was highly correlated with zinc (r = 0.90) and lead (r = 0.81) but not with copper. It is concluded that the pattern of fallout for mercury around the Flin Flon smelter is comparable to those for lead and zinc.

From theoretical considerations, Lockeretz (1974) calculated the pattern for mercury fallout at different distances from a hypothetical thermal power plant and municipal incinerator. Wet and dry fallout declined with distance from each point source in negative exponential fashion:

$$\text{Fallout} = a\ (\text{Distance})^{b}$$

The exponents (b) calculated from Lockeretz' data are given in Table 41. Fallout patterns around Flin Flon also fitted negative exponential curves. The exponents of curves fitted to data for a variety of receptors are given in Table 42. It should be noted that because of the sample designs of these studies, the curves are biased toward fallout in specific directions from the smelter (dominantly toward the east and southeast). Despite differences among metals and receptors, the exponents in Table 42 are not dissimilar to those determined for the theoretical mercury study by Lockeretz.

The exponent of these curves controls the rate of decline in fallout with distance and therefore the potential spread of contaminants away from the smelter. The median exponent in Table 42 is -1.51 and most fall within the range -1.10 to -1.70. In Figure 18, relative concentrations at different distances from the smelter are shown, assuming a concentration of 1.0 at the smelter and decay exponents of -1.10, -1.40, and -1.70. Mercury levels in soil organic layers around Flin Flon are about two orders of magnitude greater than values found elsewhere and mercury concentration in snow near Flin Flon suggests a similar difference. From Figure 18,

TABLE 41

Exponents for Curves Fitted to Theoretically Calculated Atmospheric Mercury Deposition Rates vs. Distance from Point-source (assuming complete removal)*

SOURCE	WET DEPOSITION	TOTAL DEPOSITION
Incinerator	-1.56	-1.22
Power Plant	-1.28	-1.09

*Calculated from data in Lockeretz (1974)

TABLE 42

Exponents for Curves of Metal Content vs. Distance in Various Components of the Environment around Flin Flon

METAL	PRECIP.[1]	SNOW[1]		SOIL ORGANIC LAYER[2]		BLUEBERRY STEM[2]		LAKE WATER[3]	SOIL ORGANIC LAYER[4]
	1976-77	1975-76	1976-77	1980	1981	1980	1981		
Zn	-1.73	-1.93	-1.65	-	-	-	-	-2.13	-1.68
Cd	-1.37	-3.53	-1.53	-	-	-	-	-	-
Pb	-1.13	-4.69	-1.13	-1.04	-1.24	-1.24	-0.83	-	-1.18
As	-1.67	-1.29	-1.06	-	-	-	-	-	-
Cu	-1.52	-4.98	-1.50	-	-	-	-	-1.01	-1.57
Hg	-	-	-	-1.44	-1.55	-1.14	-1.52	-	-

Sources of Data:
1 - Franzin et al., (1979)
2 - McEachern and Phillips (1983)
3 - Van Loon and Beamish (1977)
4 - Hogan and Wotton, 1984

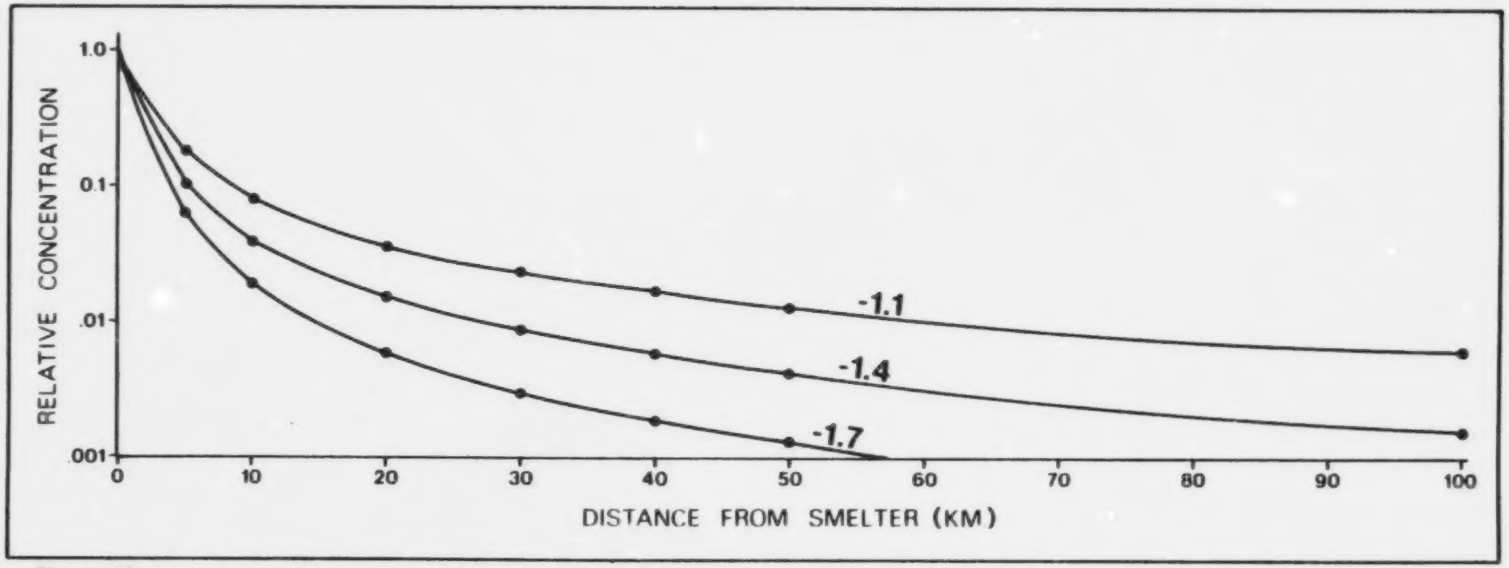


Figure 18: Relative concentrations of mercury in environment vs. distance from Flin Flon smelter for three exponents.

concentrations two orders of magnitude lower than those near the smelter occur within a few tens of kilometers, depending on the exponent assumed. Thus it appears that the significant direct impact of the Flin Flon smelter on mercury levels in the surrounding environment is limited to a radius of a few tens of kilometers.

Mercury fallout around the Thompson smelter has not been measured but elevated nickel, copper and iron levels have been recorded in vegetation and soil surface organic layer (Wotton and Hogan, 1981). Nickel and copper concentrations were highly correlated with distance and, as at Flin Flon, declined away from the smelter in exponential fashion. Wotton and Hogan (1981) concluded

> metal deposition in the Thompson area is occurring to a distance of 30 km from the smelter source. (p. 86)

supporting the results of the Flin Flon analysis. Absolute levels of mercury are probably much smaller than at Flin Flon because, as was noted above, nickel ores generally contain lower concentrations of mercury (Environment Canada, 1983).

Another factor limiting the impact of these smelters on the study region is their location with respect to prevailing wind direction. Wind roses for Flin Flon and Thompson are given in Figure 16. At Flin Flon, the dominant winds (occuring 47% of the time) are from the north, northeast and northwest, or away from the study region. Winds capable of transporting mercury to the region (from the south, southwest, and west) occur only 29% of the time (27% when frequency is weighted by average wind speed from each direction). At Thompson, winds from any direction might deposit mercury over some part of the study region but most of the region lies downwind of winds from the east, southeast or south and such winds occur for only about 22% of the time (or 20% on a velocity-weighted basis).

Thus, there is no indication that the potential effects of either smelter on mercury levels in the surrounding environment extend beyond 30-50 kilometers. Concentric circles drawn around Flin Flon and Thompson at 50 km intervals (Figure 16) suggest that the portion of the study region liable to experience mercury fallout is relatively small.

3.3.4 - Lake Sediments

Brouzes et al. (1977) have questioned the relevance of lake sediment mercury concentrations to those observed in the aquatic food chain:

> It appears that elevated mercury concentrations in sediments per se have relatively little effect on the mercury content of fish unless it is present in a soluble or solubilized form or in waters with little or no flow...(because of) the lack of evidence of any correlation between the mercury content of fish and the mercury content of sediments in northwest Quebec together with the mounting evidence of a lack of correlation existing elsewhere. (p. 20-21)

For example, lake bottom sediments in the American Falls Reservoir, Idaho, contained mercury concentrations of 210-950 ppb (mean 0.420 ppb) and lake water up to 1.78 ug/L, values respectively one and two orders of magnitude greater than those found in the study region (Kent and Johnson, 1979). Mean concentrations in fish, however, were lower than those in most of the lakes in northern Manitoba (species differences preclude direct comparison). In contrast, in Lake Jocassee Reservoir, South Carolina, where lake sediments contained only 30-40 ppb, mercury concentrations in four size classes of smallmouth bass ranged from 1.87 to 4.49 ug/g (Abernathy and Cumbie, 1977). Many other examples could be cited to demonstrate the lack of correlation between fish and lake sediment mercury levels.

Within the study region, relatively low fish mercury concentrations occur in several lakes in which lake sediment mercury levels have been greatly elevated by drainage from mine tailings (Part 3.3.2); conversely, the elevated fish mercury concentrations in the Churchill River Diversion - Rat River - Burntwood River system are associated with normal or at most, very slightly elevated lake sediment mercury contents. The scattergram of mercury concentrations in fish vs. lake sediments in the study region, given in Figure 19, supports the contention by Brouzes et al. (1977) quoted above that lake sediment mercury concentration, by itself, provides little information about the probable fish mercury content.

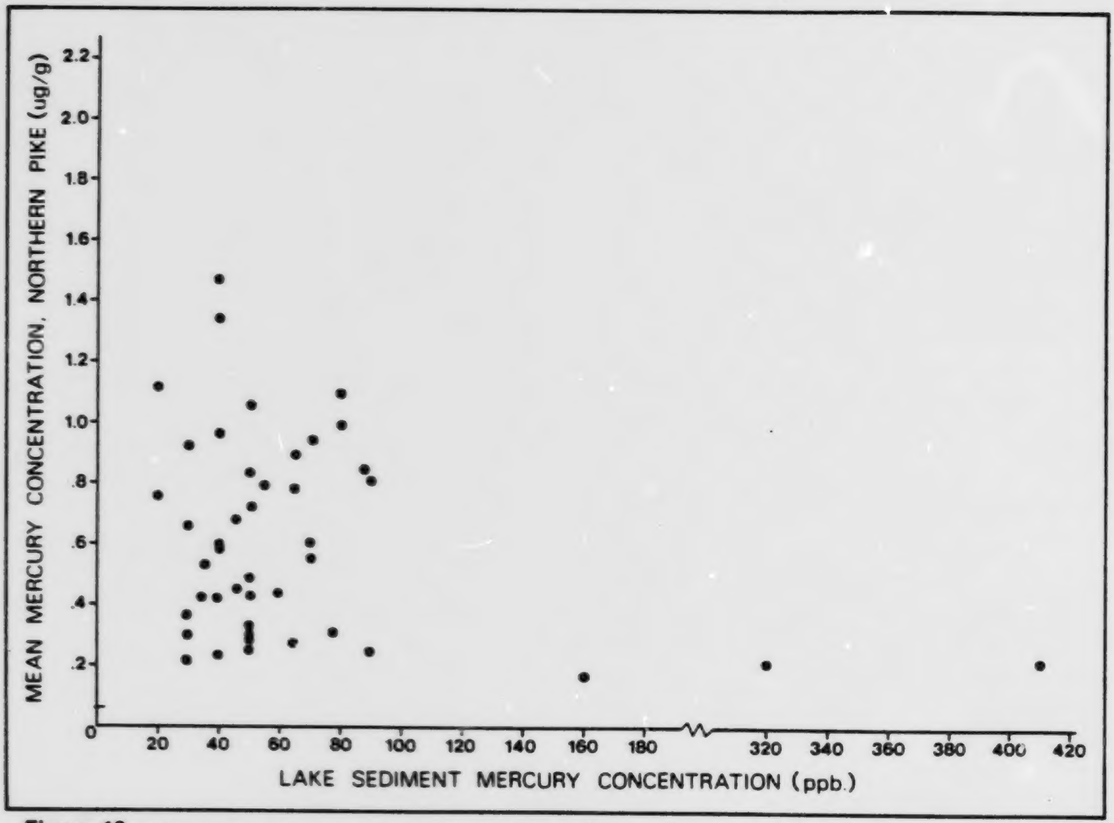


Figure 19: Mean mercury concentrations in northern pike vs. lake sediments.

PART FOUR

RELATED ENVIRONMENTAL FACTORS

4.1 - IMPOUNDMENT AND BANK EROSION

Elevated fish mercury levels are commonly found following impoundment of reservoirs. Sufficient examples have accumulated in the literature during the past decade to leave little doubt that reservoir formation is the principal cause of the increase. Mercury is introduced to the aquatic food chain by accelerated microbial activity and methyl mercury production in the inundated soil (Meister et al., 1979; Jackson, 1984) and, possibly to a lesser extent, by the direct input of mercury contained in eroded shoreline materials. Thus the relative magnitudes of the inundation ratio (ratio of flooded area to post-impoundment area) and shoreline erosion should be important factors in the post-impoundment mercury levels observed in fish.

Within the study region, the elevated fish mercury levels in lakes and rivers affected by the Churchill River diversion have been attributed to the flooding of Southern Indian Lake and Notigi Reservoir (Bodaly and Hecky, 1979; Bodaly et al., 1984). In support of this, Bodaly et al. (1984) noted in several of the lakes along the diversion route, the increase in mean mercury concentrations from levels well below the domestic marketing standard (0.5 ug/g) to levels frequently near to or in excess of the export standard (1.0 ug/g). Elevated levels appeared within two to three years of impoundment and have remained high. Since no other significant source of increased mercury flux is known, and in view of similar increases in reservoirs elsewhere, the conclusion that impoundment produced a chain reaction leading to high fish mercury levels seems fully justified. However, a recent analysis of mercury data from Southern Indian Lake (Bodaly, 1984) has revealed a trend toward decreasing concentrations in whitefish from 1978 to 1982, in apparent agreement with the conclusion by Abernathy and Cumbie (1977) that

> elevated mercury levels in fish of new reservoirs is probably transitory, declining within three to five years after reservoir filling. (p. 601).

Bodaly et al. (1984) noted that the highest mercury levels following diversion were in the Notigi Reservoir which increased in area from c. 150 km^2 (combined area of preimpoundment lakes) to 584 km^2 (Newbury et al., 1984) and concluded that increased bacterial methylation of mercury in the flooded soils was the principal cause of the increased fish mercury levels:

> The primary source of mercury was probably the upper, organic soil horizon because mercury levels in this layer were much higher than in inorganic subsoil layers. Inorganic subsoil is apparently not acting as a major source of mercury because the addition of large amounts of inorganic subsoil to the water column through soil erosion in Southern Indian Lake... did not result in fish mercury levels approaching those found in Notigi Reservoir where shoreline erosion was negligible but increase in area flooded was greater. (p. 690).

This conclusion does not, however, take into account the much greater volume of Southern Indian Lake. Relatively high mercury concentrations measured in suspended sediments (0.255 ug/g, Bodaly et al., 1984) suggest that shoreline erosion around Southern Indian Lake introduced significant mercury to the lake, but no conclusion can be drawn concerning its form or contribution to fish mercury levels.

Lakes downstream of the Notigi Reservoir also exhibited high post-impoundment mercury levels in fish (Bodaly et al., 1984). In Wuskwatim Lake, average mercury concentration in walleye increased from 0.34 ug/g in 1970-76 to 0.88 ug/g from 1979 to 1981. Only post-impoundment data were available for the other lakes but most had concentrations greater than 1.0 ug/g. Although not directly impounded, this portion of the diversion route experienced higher water levels due to increased flow in the Rat-Burntwood Rivers.

For most of the lakes along this system, preimpoundment data are sparse or non-existent. Fish in many of the lakes adjacent to the Diversion route, but unaffected by it, also exhibit comparatively high mercury concentrations (Table 43). Thus, while the conclusion that a rise in mercury concentration has occurred due to impoundment is undoubtedly correct, other factors seem capable of contributing to high concentrations in this part of the study region and the extent of rise due to impoundment alone is unclear.

TABLE 43

Mercury Concentrations in Northern Pike in Lakes Adjacent to the Churchill Diversion

LAKE	SURVEY TYPE	MEAN (ug/g)	RANGE
Apeganau	Commercial	0.91	0.84-1.03
Cousins	Commercial	0.51	-
Leftrook	Commercial	0.29	0.25-0.33
	Lake Survey	0.27	0.10-0.40
Macheewin	Commercial	0.62	0.33-0.98
	Lake Survey	0.70	0.29-1.78
Osik	Commercial	1.57	1.32-1.83
	Lake Survey	1.46	0.41-2.21
Rusty	Commercial	0.50	-
	Lake Survey	0.56	0.35-0.72
Suwanee	Commercial	0.29	0.17-0.52
	Lake Survey	0.17	0.13-0.23

Mercury levels in northern pike from other impounded lakes in the study region are given in Table 44. Approximately 26 km2 of shoreline along Sipiwesk Lake were flooded for the Kelsey Generating Station in 1960, producing considerable erosion (Teillet, 1979). Stephens and Gull Lakes, created by closure of the Kettle Rapids Dam in 1970-71, flooded 325 km2 and caused significant bank erosion (Teillet, 1979). Reindeer Lake was raised about 2 m by the construction of the Rocky Falls and Whitesand Dams in Saskatchewan in 1938 and 1942, flooding 26 km2 (Teillet, 1979). Sokatisewin Lake, also in Saskatchewan, was impounded behind the Island Falls Dam in 1928 to provide electricity for the smelters at Flin Flon (Hecky, 1975). Laurie River has had several structures placed across it to impound water since the early 1950's. Fish mercury data are not available for lakes directly affected by the Hydro structures but Tod and Laurie Lakes have been impounded behind a control structure upstream. No data on the extent of flooding or the amount of shoreline erosion on these lakes are available. Examination of maps suggests that it was relatively small.

The exceptionally high concentration of mercury in fish from Stephens and Gull Lakes (Table 44) is significant since these lakes were impounded most recently, had very high inundation ratios and produced very large amounts of shoreline erosion (Teillet, 1979). Furthermore, lakes adjacent to Stephens Lake have much lower mean concentrations (Butnau - 0.32 ug/g; Limestone - 0.51 ug/g) and a 1971 commercial fish sample of Moose Lake, now flooded by Stephens Lake, indicated a mean concentration of only 0.20 ug/g with a narrow

TABLE 44

Mercury Concentrations in Northern Pike for Impounded Lakes (other than Churchill Diversion Lakes)

LAKE	SURVEY TYPE	MEAN (ug/g)	RANGE
Sipiwesk	Commercial	0.92	0.16-2.31
	Lake Survey	0.81	0.07-3.55
Stephens	Lake Survey	1.41	0.37-3.66
Gull	Lake Survey	1.29	0.72-1.76
Tod	Commercial	1.47	-
Laurie	Commercial	1.06	0.69-1.54
	Lake Survey	1.12	0.75-2.25
Reindeer (Man)	Commercial	0.36	0.04-0.69
	Lake Survey	0.63	0.42-1.02
Reindeer (Sask)	Commercial	0.39	0.18-0.62
	Lake Survey	0.52	0.35-1.07
Sokatisewin	Commercial	0.56	0.32-1.18

range (0.15-0.27 ug/g). The concentrations in Sipiwesk Lake are somewhat lower (although still high in absolute terms), possibly because impoundment occurred a decade earlier and flooded a smaller area. The still lower values for the Saskatchewan impoundments presumably also reflect the age of the reservoirs and the relatively small inundated area. Mercury concentrations are also extremely high on Tod and Laurie Lakes. These may be related to impoundment but sulphide mineralization at nearby Fox Lake Mine offers an alternative (or additional) explanation.

Thus the mercury concentrations in the lakes listed in Table 44 support the suggestion by Bodaly et al. (1984) that inundation accounts for some of the observed high mercury concentrations in the study region and are in general agreement with the observation by Abernathy and Cumbie (1977) that mercury concentration declines with reservoir age.

4.2 - ENVIRONMENTAL ACIDITY

The pH of lake water has been identified as an important factor in the formation of monomethyl mercury and its uptake by fish (Brouzes et al., 1977). Fagerstrom and Jernelov (1972) showed that at a pH of about 7.8, mono- and dimethyl mercury formed in approximately equal amounts. As pH was lowered, the proportion of monomethyl mercury increased until at pH 6.0 virtually only monomethyl mercury formed. Since monomethyl mercury is the form in which mercury is normally assimilated by fish, lakes with lower pH may be expected to exhibit higher fish mercury concentrations, other things being equal. Monomethyl mercury can also be formed biologically and abiologically in soils with low pH (Rogers, 1977).

The average pH of several NAQUADAT stations in the study region is given in Table 45. These values appear to be typical of other major water bodies in the study region for which published data are available, even those receiving acidic drainage from mine tailings (Part 3.3.2). Thus it may be concluded that the normal pH of most major lakes and rivers is greater than 7.0, in some cases significantly so. Similar values were reported by Shewchuk (1983) throughout much of northern Saskatchewan. In northwestern Quebec, Brouzes et al. (1977) observed that very low pH (3.8-5.5) in the snow spring freshet caused a drop in pH of the Bell River from 6.5 to 5.5. Shewchuk (1983) obtained snow pH values ranging from 4.9 to 7.9 for snowmelt in the Flin Flon area and Wotton and Hogan (1981) reported an average snow pH of 4.8 at thirteen sites near Thompson

TABLE 45

pH of Selected Surface Waters in the Study Region (NAQUADAT)

		n	MEAN	RANGE
Burntwood River at Thompson	-1974	20	7.7	7.2-8.0
	-1975	7	7.6	7.4-7.9
	-1976	8	7.8	7.5-7.9
	-1977	7	7.6	7.1-7.9
Churchill River above Granville Falls	-1974	6	7.6	7.2-7.9
	-1975	5	7.6	7.4-7.8
	-1976	4	7.7	7.4-7.9
	-1977	4	7.2	6.8-7.6
Southern Indian Lake Above Missi Falls	-1974	3	7.5	7.4-7.7
	-1975	5	7.6	7.4-7.8
	-1977	6	7.7	7.4-8.0
Nelson River at Kelsey Generating Station	-1974	10	8.3	7.9-8.7
	-1975	4	7.7	7.1-8.1
	-1976	3	8.0	7.8-8.2
	-1977	5	7.8	7.4-8.0
Barrington River West of Opachuanau Lake	-1974	5	7.4	7.3-7.6
Taylor River near Thompson	-1974	6	7.5	7.1-7.9
Rat River above Threepoint Lake	-1974	3	7.6	7.4-7.9

in 1980 and 1981, a value well below the normal pH of precipitation. Perhaps surprisingly, no clear correlation was observed between pH and distance from smelter at either Flin Flon or Thompson, at least within the distances covered by these studies. No consistent drop in pH during the spring freshet is apparent in the limited data available for the lakes and rivers in the study region and water bodies of significant size appear to have a good buffer capacity.

Nevertheless, acidic conditions do exist within the study region. The distribution of pH values in waters sampled during the Geochemical Reconnaissance Survey of NTS Map 64C is given in Table 46. The most acidic water bodies were small, especially those classifed as "ponds" (less than 0.25 km^2). Lakes larger than 5 km^2 (the largest category listed in the data summary) had higher pH but a significant number exhibited acidic conditions liable to enhance the formation of monomethyl mercury.

There is no evidence, however, that mercury concentrations are in fact related to lake water acidity in the study region. Lake pH and mercury concentrations in commercial fish samples from 29 lakes on NTS Map 64C are tabulated in Table 47 and plotted on Figure 20. No pattern of association is evident in either, despite a wide range in pH (6.55-7.55) and mercury concentration (0.22-1.47 ug/g). Clearly other factors are capable of mitigating whatever singular effect lake water acidity may have.

TABLE 46

pH of Waters Sampled in National Geochemical Reconnaissance Survey, NTS Map 64C (Geological Survey of Canada, 1983)

pH CLASS	% OF ALL SAMPLES		
	ALL SAMPLES	PONDS <0.25 km^2	LAKES >0.5 km^2
4.8-5.0	0.2	0.5	0
5.1-5.3	0.1	0	0
5.4-5.6	0.2	0	0
5.7-5.9	0.9	2.0	0
6.0-6.2	3.7	6.9	2.1
6.3-6.5	11.5	14.8	7.0
6.6-6.8	25.4	26.1	18.5
6.9-7.1	32.8	28.1	42.0
7.2-7.4	18.9	16.7	25.5
7.5-7.7	5.9	4.9	5.9
7.8-8.0	0.4	0	0
n	1364	203	286

TABLE 47

Mean Mercury Concentrations in Commercial Samples of Northern Pike, and Lake pH on NTS Map 64C

MERCURY CONCENTRATION (ug/g)	pH		
	6.1-6.5	6.6-7.0	>7.1
≤0.24	0	2	0
0.25-0.54	0	4	7
0.55-0.84	0	4	4
0.85-1.14	0	3	4
≥1.15	0	1	0

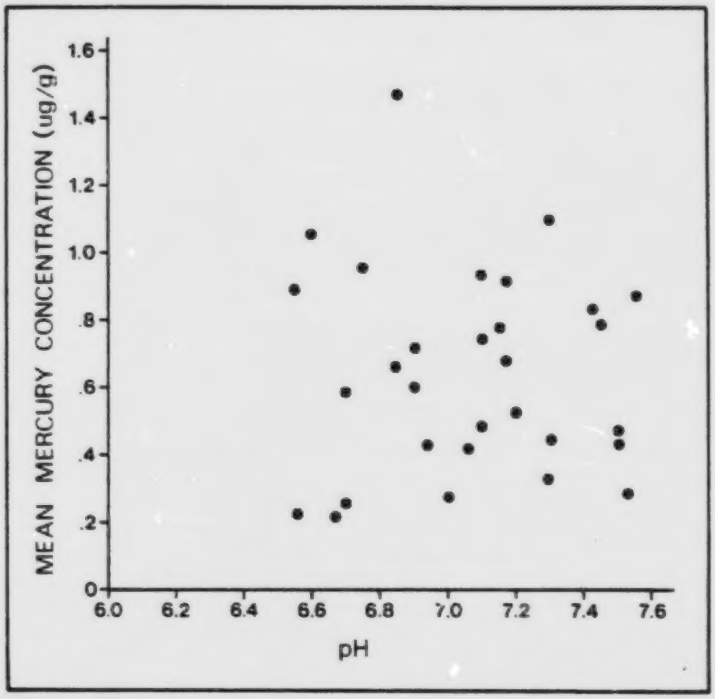


Figure 20: Mean mercury concentration in northern pike vs. lake pH, NTS Map 64C.

Data on soil pH are limited but acidic conditions are probably common throughout the study region. Values as low as 3.4 were measured in the soil surface organic layer near Thompson (Wotton and Hogan, 1981) and Flin Flon (Hogan and Wotton, 1980) and pH's less than 5.0 were normal in each area. Again no correlation between pH and distance from the smelter was found within the 30-40 km radii of each study, despite the fact that the Thompson and Flin Flon smelters are, respectively, the third and fourth largest sources of sulphur dioxide emission in Canada (House of Commons Subcommittee on Acid Rain, 1981).

The acidity observed in snow, small water bodies, bogs and surface soil layers suggests that conditions suitable for monomethyl mercury formation are widespread in the study region. Waters draining these areas may contain biologically significant quantities of this toxic form of mercury, although possibly in amounts which are below the normal detection level when diluted in large water bodies. Uptake of this monomethyl mercury may be highly seasonal with the result that neither the acidity nor the mercury are detected in normal measurement programmes. Shoreline erosion and inundation of low-lying areas during reservoir formation or possibly even during normal spring flood events would accelerate the process. In fact, any process which increases the "links" between fish and areas of low pH and rapid monomethyl formation may produce elevated mercury levels. For example, artificial drainage of (probably acidic) peat bogs in northern Finland appears to have produced accelerated accumulation of mercury in lake sediments

(Simola and Ladenius, 1982). To demonstrate that such processes are important in the study region would require much more extensive data than are presently available.

4.3 - FOREST FIRES

Forest fires may be expected to mobilize mercury contained on and in the biomass and redistribute it into the atmosphere either as vapour or attached to particulates. The latter form is probably of greater importance in the present context because of its more concentrated fallout pattern. Mercury is readily adsorbed onto particulate matter and forest fires are the largest natural contributor to atmospheric particulate loading in Manitoba (Environment Canada, 1983). Following a major fire, runoff and sediment yield normally increase for a variable period, depending upon the nature of the fire, substrate and proportion of the watershed burned. Further redistribution of mercury in the soil can be anticipated. Other nutrients contained on the burn products and in the post-fire runoff and sediment may stimulate production of monomethyl mercury in the bottom sediments and/or water column.

The amount of mercury released annually to the atmosphere from fires in the boreal forest region of Manitoba was estimated by the Air Pollution Control Directorate to be only 0.02% of the provincial total emitted annually from natural sources, based on the volume of commercially exploitable timber lost to fire and a mean mercury content of the biomass of 0.4 ug/g (Environment Canada, 1983).

This estimate, however, has little relevance in evaluating the impact of fire on an adjacent water body. A better perspective on the local effect of forest fires may be obtained by comparing the potential amounts of mercury released per unit area from fire with those produced by runoff and shoreline erosion. The bases for these calculations are summarized in Table 48. One square kilometer of forest fire might release 4 times as much mercury as shoreline erosion or 500 times as much as runoff from equivalent areas. Runoff and shoreline erosion, however, concentrate mercury in the receiving water body whereas particulate fallout from a forest fire is dispersed over an unknown but much larger area. In addition, the value given in Table 48 for shoreline erosion does not take into account direct methyl mercury input from the underlying inundated soil. These differences are mitigated somewhat by the differences in time base among the events. The runoff and shoreline erosion figures are annual inputs whereas the mercury fallout from most fires would occur over a period of days or weeks - i.e. one to two orders of magnitude greater in intensity. Thus the fire input would be more analogous to a pulse of mercury. The other limitations of these calculations are obvious and need no further elaboration. Nevertheless, they do suggest that significant quantities of mercury may be mobilized during large fires.

Forest fires are a normal component of boreal ecology with "natural" return periods of the order of 100 years (Rowe and Scotter, 1973). The period of intensive mining exploration has

TABLE 48

Calculated Mercury Release from Forest Fire, Runoff and Shoreline Erosion

FOREST FIRES

Assumptions

- Boreal biomass - 25 kg/m^2 (Seiler and Crutzen, 1980)
- Mercury content of biomass - 0.4 ug/g = 0.4 mg/kg (Environment Canada, 1981)
- Burning efficiency - 20% (Seiler and Crutzen, 1980)

Amount of mercury released

25 kg/m2 x 0.4 mg/kg x 0.2 = 2 mg/m2 = 2 kg/km2

RUNOFF

Assumptions

- Average annual runoff - 0.2 m = 0.2 x 106 m3/km2
- Average Mercury content of runoff - 0.02 ug/L = 0.02 mg/m3

Amount of mercury released

0.2 x 106 m3/km2 x 0.02 mg/m3 = 4 g/km2 = 0.004 kg/km2

SHORELINE EROSION ON SOUTHERN INDIAN LAKE

Assumptions

- Average annual erosion, 1976-78 - 4 x 109 kg (Newbury and McCullough, 1984)
- Average mercury content of eroded materials - 0.05 ug/g = 0.05 mg/kg
- Flooded area - 414 km2 (Newbury et al., 1984)

Amount of mercury released

4 x 109 kg x 0.05 mg/kg ÷ 414 km2 = 0.48 kg/km2 = 0.5 kg/km2

increased fire frequency considerably. In northern Saskatchewan Scotter (quoted in Kelsall, 1968) found

> the acreage burned per year by forest fires had increased 1.5 times in the last 14 years compared to the previous 60 years and 3.5 times when compared with the period extending from 1840 to 1844. (p. 266)

In northern Manitoba, Samuel Hearne noted the great extent of burned area 200 years ago. Major fires have occurred within the study region in recent decades (Figure 21). Especially noteworthy are the large burns along the Churchill Diversion-Burntwood River system and in the area southeast of Lynn Lake since these areas contain most of the lakes in which very high fish mercury concentrations have been observed. Examples of large burns exceeding 100 km^2 in two areas near the Churchill Diversion are shown in Figure 22.

Little is known about the effects of forest fires on the redistribution of minor elements in the ecosystem and nothing at all about the fate of mercury. Charcoal layers in lake sediments have been used to calculate prehistoric fire frequency (eg. Swain, 1973; Terasmae and Weeks, 1979), and indicate that burn products reach nearby lakes in recognizable amounts. Among the conclusions reached by Wright and Heinselman (1973) from papers presented at a symposium on the ecological role of fire in natural conifer forests in North America were the following:

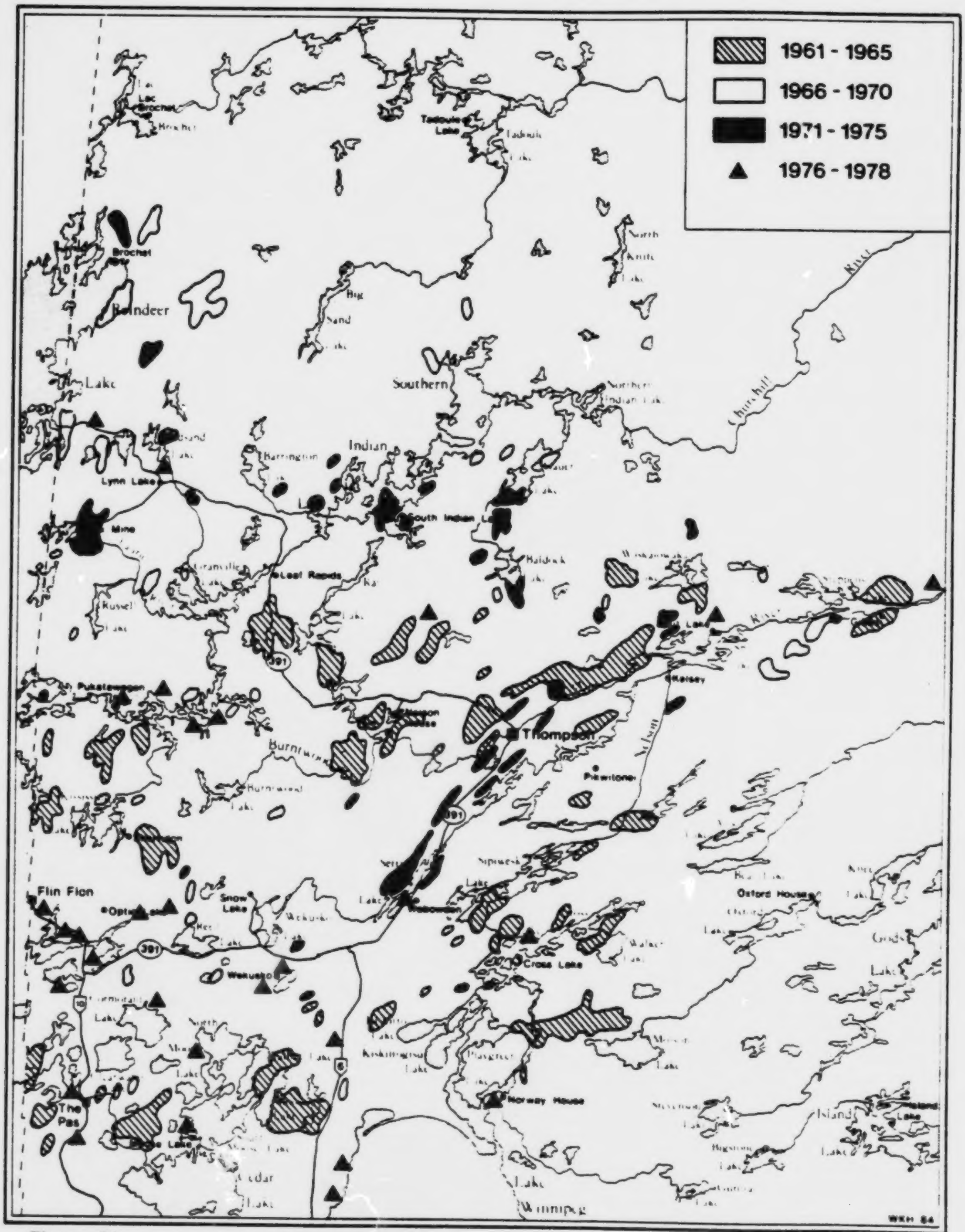

▨ 1961 - 1965
☐ 1966 - 1970
■ 1971 - 1975
▲ 1976 - 1978
Lac Brochet
Brochet
Tadoule Lake
Tadoule Lake
North Knife Lake
Brochet
Reindeer
Lake
Big Sand Lake
Southern
Indian
Northern Indian Lake
Churchill
River
Barrington Lake
Lynn Lake
Mine
South Indian Lake
Granville Lake
Leaf Rapids
Russell Lake
Baldock Lake
Pukatawagan
Burntwood
Burntwood Lake
Thompson
Kelsey
Pikwitonei
Nelson
Sipiwesk
Lake
Wabowden
Flin Flon
Snow Lake
Wekusko
Reed Lake
Wekusko
Cross Lake
Walker Lake
Oxford House
Oxford Lake
Gods Lake
North
Lake
Kiskittogisu Lake
Playgreen Lake
Norway House
Molson Lake
The Pas
Moose Lake
Cedar Lake
Lake
Winnipeg
Stevenson Lake
Island Lake
Bigstone Lake
391
10
6
WKH 84

Figure 21.: Major forest fires in study region, 1961-78 (redrawn after Teillet, 1979).

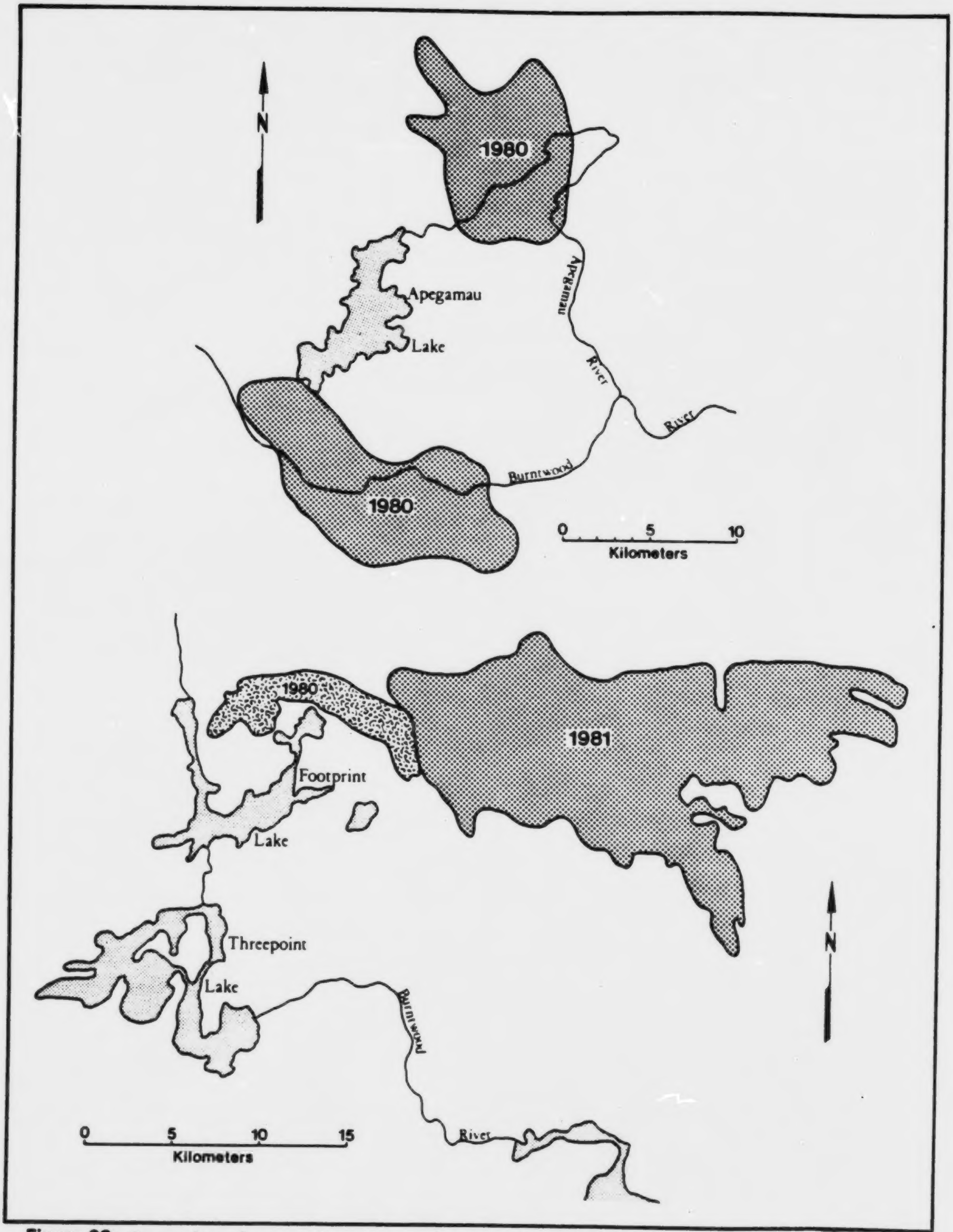

N
1980
Apegamau Lake
Apegamau River
Burntwood River
1980
0 5 10
Kilometers
1980
1981
Footprint Lake
Threepoint Lake
Burntwood River
N
0 5 10 15
Kilometers

Figure 22: Examples of recent burns near the Churchill Diversion.

- Fires directly release the mineral elements as ash from the living and dead organic substances burned.
- Fires may indirectly release still more mineral elements through increased decomposition rates of remaining organic layers and other remains, leaching or erosion of mineral soils, physical spalling of rocks and the subsequent breakdown of rock fragments etc. The quantities involved in (the above) are poorly known. They are certainly significant if the fires are intense and frequent. For example, the exfoliation of granite boulders and rock outcrops heated intensively by the burning of moss and lichens and other fuels, as manifested in the Little Sioux fire of 1971 in northern Minnesota, may be far the most important rock weathering process in the region.
- Fires directly recycle the carbon of herbs and shrubs and the foliage, bark and wood of trees.
- In fire-dependent ecosystems, fire is a major factor in nutrient cycling...Rates and pathways are influenced by fire frequency and intensity.
- Fire recycles nutrients that otherwise might not move.

Detailed data on burn areas, the nature of the fires, wind directions with respect to adjacent water bodies, the mercury content of major vegetative components etc. would be required to demonstrate a correspondence in time and space between forest fires and fish mercury levels. The calculations in Table 43, however, suggest that forest fires may play a significant role in mobilizing mercury contained in the vegetation and rocks.

PART FIVE

CONCLUSIONS

The major conclusions reached in this study are summarized below.

ENVIRONMENTAL MERCURY LEVELS

1. The volcanogenic rocks of the region contain elevated mercury concentrations with respect to both other rock types in the study region and similar rocks in other parts of Canada. Particularly elevated concentrations are associated with massive sulphide bodies rich in zinc sulphide (sphalerite).

2. Mercury concentrations in lake sediments are within the normal range for other locations and thus cannot be considered elevated.

3. The surface waters of the study region contain mercury near or below the detection limit, with no evidence of elevated concentrations.

4. The limited data available suggest that mercury concentrations in soils generally fall within the normal range for similar soils elsewhere in Canda, with locally elevated values occurring near sulphide mineralization. Highest concentrations occur in the vicinity of the smelter at Flin Flon.

5. No conclusions can be reached regarding general mercury concentrations in vegetation in the study region because of a lack of data. Very elevated concentrations occur in shrubs near Flin Flon.

6. Fish in the study region contain higher mercury levels than those of northern Saskatchewan. About 40% of the lakes have average concentrations in northern pike which exceed standards for domestic sale and 8% exceed export standards. Maximum values occur along the Churchill River Diversion - Burntwood River - Nelson River system and in the region southwest of Lynn Lake. Below average concentrations occur in lakes in the Flin Flon region and in the Superior Province east of the Nelson River. Elsewhere, high, moderately high, and average concentrations are interspersed in a generally random pattern.

POTENTIAL SOURCES OF MERCURY

7. There is no evidence to indicate that mercury is transported into the region by human or natural agencies and it is concluded that virtually all of the mercury in the ecosystem originates within the study region.

8. The ultimate source of mercury in the ecosystem of the study region is the bedrock, particularly volcanogenic rocks. A general association between elevated fish mercury concentrations and rocks with high sulphide potential is apparent.

9. The pathways by which mercury moves from the rock into the aquatic ecosystem remain undefined.

10. No point sources of any regional consequence can be identified.

11. The smelter at Flin Flon is a major source for elevated mercury concentrations in soil, snow, and vegetation but there is no evidence that its effects extend beyond a radius of a few tens of kilometers from the smelter. No mercury data exist for the smelter at Thompson but data for other metals suggest that its impact is limited to a 30 km radius and the ores processed there probably contain less mercury than those at Flin Flon.

12. Active and inactive gold and base metal mining operations have no identifiable impact on the mercury concentration of fish in nearby lakes.

13. Most of the water bodies in which the highest fish mercury concentrations have been observed have experienced inundation, increased flow and/or shoreline erosion consequent upon reservoir formation.

14. The pH of major water bodies are neutral or alkaline and no association between acidity and fish mercury concentrations can be demonstrated. Nevertheless, the possibility that environmental

acidity plays an important role in the pattern of elevated concentrations should not be ruled out.

15. Forest fires may have mobilized and redistributed mercury in the biomass and soil.

ASSESSMENT OF DATA BASE

16. Sufficient general data on the mercury content of most rock types and sulphide ores exist to evaluate the general level of mercury in the bedrock of the study region. More detailed data might be beneficial but it is doubtful if the results would justify the expense or materially improve our knowledge of mercury distribution.

17. Soil data are abundant near Southern Indian Lake but virtually non-existent throughout the rest of the study region. Because soil is the major link between the mercury-containing rock and the terrestrial and aquatic ecosystems, a broader, spatially-distributed data base reflecting the full range of environments is highly desirable.

18. Lake sediment data are abundant and will become even more so when the additional Geochemical Reconnaissance data for other map sheets in the study region are released in 1985. No further general data on the mercury content of lake sediments are required.

19. Surface water samples normally provide little information on mercury concentration and no additional sampling is warranted beyond the continuation of normal water quality sampling.

20. Data on mercury concentrations in snow samples are meagre at present. Snow surveys offer an efficient method of monitoring cumulative atmospheric fallout in winter and should be undertaken.

21. The mercury content of vegetation is the least adequately documented of all important environmental components in the study region. Shrubs near Flin Flon have been sampled but because of the proximity to the smelter, these provide no basis for evaluating mercury in vegetation elsewhere in the study area.

22. Data on mercury concentrations in fish in the study region are abundant. The commercial monitoring and lake survey programmes should be continued but no general expansion of fish mercury determinations seems necessary. However, specific recommendations regarding these programmes are given in Part 6 of this report.

PART SIX

RECOMMENDATIONS

The following recommendations arise out of the conclusions in Part Five and should be given priority in any further analysis of mercury contamination in northern Manitoba lakes.

1. Soil data collecton programme: A broad, spatially-distributed data base on overburden mercury concentrations is desirable. The term "overburden" is used here since variations in mercury concentration are more likely to be related to the characteristics of the underlying bedrock and textural variations in the surficial material rather than to strictly pedogenic processes. Thus, the sampling strategy should be designed to reflect these differences. Special attention should be paid to the effect on mercury concentration of textural variations in the drift cover produced by glacial processes.

2. Vegetation data collection programme: Virtually no data exist on general mercury levels in the vegetation of the study region. A sampling programme should be initiated, covering major cover types and plant components. Special attention should be paid to the lichen cover because lichens are known to accumulate air-borne contaminants, particularly radionucleides. The samples should be collected in conjunction with the soil sampling programme.

3. Snow sampling program: Snow offers an efficient method of monitoring mercury fallout and a sampling programme should be initiated in the spring. The samples should be broadly distributed over the study region to provide data on background levels of atmospheric fallout and potential mercury content of the spring freshet. Simultaneous samples of small streams during the freshet period should also be collected. Snow and runoff acidity and sulphate ion concentration should also be measured.

4. pH measurement programme: The pH of environmental subsystems should be monitored to determine whether acid conditions favourable to the formation of monomethyl mercury occur locally and/or seasonally.

5. Forest fire study: The potential contribution of forest fires to the mercury content of nearby lakes warrants detailed study. Such a study involves several components:

(a) Existing data on fire history near lakes with high and low fish mercury concentrations should be compiled to determine if an association exists. Particular attention should be paid to lakes in which pre- and post-fire fish data are available.

(b) Lakes near future fires should be monitored, particularly those for which pre-fire data are available. Fish mercury levels should be determined for a number of years thereafter. Other aspects of water and lake sediment chemistry should also be monitored for mercury and nutrient influx.

(c) A water and sediment sampling programme should be established near recent burns to determine the post-fire influx of mercury in the runoff and adsorbed on sediments.

(d) Sediment cores from lakes near areas with an established burn history should be examined.

6. Impoundment study: Most studies of the effects of impoundment on fish mercury levels suffer from a lack of pre-impoundment data. To correct this, a list of all planned water impoundment projects in and near the study region should be compiled and a major base-line study should be initiated in the most likely of these. The data collection programme should cover all major components of the terrestrial and aquatic ecosystem, including detailed study of the soils and vegetation in the area likely to be inundated and should continue for perhaps a decade following impoundment. Such a study clearly involves the commitment of human and financial resources for an extended period before producing definitive results. However, many of the components may be integrated within the soil and vegetation data-collection programmes recommended above and determination of fish mercury levels could be integrated into the existing lake survey programme without significant additional expense. Other necessary data on such aspects as shoreline erosion, water quality, inundation area etc. are likely to be included in environmental impact studies normally conducted on major impoundments. The importance of this

recommendation lies in the anticipation of impoundment and assignment of existing resources to compile a data-base specifically focussed on mercury. Proposed hydro-electric projects on the Churchill River in Saskatchewan, for example, offer several possible target sites where mercury data on soils and vegetation are non-existent.

7. Fish survey: As was suggested in Part Five, data on fish mercury concentrations are abundant and, with some exceptions, no general expansion of the data collection programme appears necessary to identify mercury sources within the study region. However, within the existing measurement programme, the following recommendations should be given priority:

(a) Continued monitoring of lakes with current high levels of mercury in fish is an obvious requirement, particularly in impounded lakes in view of the downward trend in mercury concentrations such lakes may display.

(b) Representative lakes in all concentration categories should be identified for annual sampling to establish a firm time-line data base.

(c) Fish in lakes near future forest fires should be monitored for a sufficient period to establish whether fires affect mercury levels.

(d) Monitoring of fish in all water bodies likely to impounded in the future should begin immediately.

(e) Because spatial relationships are of primary importance in any identification of sources of mercury, all fish mercury data should be geocoded immediately to enable rapid, efficient display of spatial patterns and to facilitate statistical analysis.

8. Shoreline characteristics study: The shoreline characteristics of a representative sample of lakes with low, intermediate and high levels of mercury in fish should be examined. Shoreline characteristics relevant to such a study are outlined below:

Characteristic	Potential Relevance
(a) Proportions of bedrock- and overburden-controlled shoreline.	Affects the amount of natural erosion and input of soil mercury.
(b) Texture and lithology of shore materials.	Affects the amount of erosion, mercury content and buffer capacity of soils.
(c) Proportion of actively eroding shoreline.	See (a) and (b) above.

(d) Lake margin topography.	Affects area liable to be periodically inundated by natural fluctuations in water level.
(e) Lake bathymetry.	Extensive shallow (marsh or bog) areas subject to seasonal inundation may be sites of monomethyl mercury formation.
(f) Lake level regime.	Affects frequency and extent of periodic flooding of shoreline by natural processes which may produce effects analogous to artifical impoundment.

Systems of shoreline classification devised for impoundment impact studies in northern Manitoba (Newbury et al., 1973) and Saskatchewan (Muir et al., 1975) may be readily adapted to such a purpose and much of the data may be obtained from large-scale aerial photographs.

9. Statistical Analysis of Lake Sediment Mercury Concentration Data: A large body of data on the mercury content of lake sediments is available and this will increase by about threefold in May-June, 1985, when the results of the National Geochemical Reconnaissance Survey of NTS Maps 64B, 64F, and 64G are released. Such data may provide insight into the spatial patterns of mercury derived from bedrock sources and, thus, of the general availability of mercury to fish; these patterns may then suggest other factors which govern the uptake of mercury by fish. Consequently, a major statistical analysis of the entire lake sediment survey data body is recommended, with the identification of spatial patterns and associations as its principal objective.

Univariate statistics may be used to establish criteria to characterize concentrations as elevated, normal, or low. Standardized data (e.g. as standard scores) may then be mapped to identify regions of anomalous concentrations. A two-dimensional polynomial trend surface analysis may assist in defining broad patterns of lake sediment mercury contents. Once the univariate analysis is complete, a multivariate analysis of all lake sediment characteristics reported in the Geochemical Reconnaissance Survey programme should be undertaken. Preferred associations among mercury and other metals might be anticipated since the mercury contents of various sulphides differ, and such sediment or water characteristics as pH and loss-on-ignition may also be important in "explaining" the distribution of mercury. Factor analysis should be conducted on the entire data body to identify such joint associations as may exist. Mapping and trend surface analysis of the factor scores may then lead to further clarification of the controls on the distribution of mercury. Because bedrock is the principal source of the mercury, lithology should be quantified using a "dummy variable" code. Variations in lake sediment mercury due to rock type may then be tested by regression analysis and analysis of variance.

While the data body is large, all National Geochemical Reconnaissance data are available in computer tape format and statistical analyses of the type recommended above are relatively straight-forward and geocoded locations of all measurement sites permit efficient mapping of the results.

10. Integration and Scheduling of studies: The studies recommended above should not be conducted independently. Surveys of vegetation and soil mercury concentrations clearly need to be conducted simultaneously with a common sampling strategy. These may provide much of the data required for the forest fire and inundation studies, if the latter are taken into account in designing the survey. Some consideraton might be given to establishing a small representative watershed within which coordinated data on soil, vegetation, acidity and runoff data could assist in identifying the pathways by which mercury moves from the underlying bedrock, through the terrestrial, and into the aquatic ecosystem.

Some of the studies recommended above could be undertaken and completed in the spring of 1985; others require field work which could be completed in the summer of 1985, while still others may begin in 1985 but require commitment over a period of years to provide reliable answers to the questions raised. Realistic time-periods for the major recommendations are summarized below:

SPRING, 1985	SUMMER-FALL, 1985	CONTINUING BEYOND 1985
Snow sampling		
Freshet runoff sampling		
pH measurement		
Geocoding of fish data		
	----- Statistical analysis -----	
	----- Shoreline analysis -------	
------------- Forest fire studies	------------------------------	------------------------------
	--- Soil sampling --------------	
	--- Vegetation sampling --------	
	----------- Fish monitoring ----	-------------------
	----------- Impoundment studies	---------------

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APPENDIX I

LIST OF PERSONS CONTACTED DURING THE PREPARATION OF THE REPORT

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Mr. C. B. Orcutt, Environmental Control Services, Manitoba Department of Environment and Workplace Safety and Health, Winnipeg.

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APPENDIX II

SUMMARY OF MERCURY ANALYSES IN FISH FROM WATER BODIES IN THE STUDY AREA

Notes

1. Location: Unless specified otherwise, all water bodies are lakes. Numbers under each name provide the township/range grid reference.

2. Species:
   - NP - northern pike
   - YP - yellow pickerel (walleye)
   - WF - lake whitefish

3. Survey Type:
   - LS - Lake survey
   - CM - Commercial monitoring programme

LOCATION	SPECIES	SURVEY TYPE	YEARS	N	CONCENTRATION (ug/g)		% GREATER THAN	
					MEAN	RANGE	0.54	1.00
Aiken River 82-83/9-10E	NP	LS	1982	14	0.81	0.16-1.14	86	36
	YP	LS	1982	7	0.70	0.45-1.06	86	14
Allen 82/20 W Churchill River South of Granville F	NP	CM	1978, 1981	4	0.48	0.34-0.61	25	0
		LS	1978, 1981	20	0.47	0.24-0.84	35	0
	YP	CM	1978, 1980 1981	5	0.39	0.30-0.50	34	0
Amy 87/22 W	NP	CM	1982	4	0.88	0.41-1.28	75	25
		LS	1982	20	0.70	0.34-1.48	55	25
	YP	CM	1982	4	0.60	0.47-0.76	75	0
		LS	1982	21	0.63	0.34-1.01	57	5
Angle 75/28 W	YP	CM	1971	2	0.19	0.14-0.24	0	0
Apeganau 76/14 W	NP	CM	1976, 1977	3	0.91	0.84-1.03	100	33
	YP	CM	1976, 1977	8	0.47	0.31-0.63	38	0
Apussigamasi 79/1-2W	NP	CM	1982, 1983	3	1.34	0.66-1.84	100	67
		LS	1982, 1983	15	1.25	0.39-2.49	93	73
	YP	CM	1982, 1983	2	2.06	1.75-2.37	100	100
		LS	1982, 1983	11	2.10	1.36-2.90	100	100
Archibald 76/8E	NP	CM	1983	1	0.52	0.52-0.52	0	0
		LS	1983	5	0.47	0.26-0.66	20	0
	YP	CM	1983	1	0.17	0.17-0.17	0	0
		LS	1983	6	0.16	0.11-0.34	0	0
Armstrong 77-78/4E	NP	CM	1971, 73-76 1979, 1980	8	0.22	0.18-0.36	0	0
		LS	1979, 1980	10	0.27	0.13-0.51	0	0
	YP	CM	1971, 73-76 1979, 1980	9	0.16	0.11-0.23	0	0
		LS	1979, 1980	11	0.25	0.12-0.39	0	0

LOCATION	SPECIES	SURVEY TYPE	YEARS	N	CONCENTRATION (ug/g)		% GREATER THAN	
					MEAN	RANGE	0.54	1.00
Ashley 105/3-4 W	NP	CM	1983	1	0.26	0.26-0.26	0	0
		LS	1983	5	0.25	0.18-0.34	0	0
Assean 83/6-7 E	NP	CM	1982	1	0.57	0.57-0.57	100	0
		LS	1981, 1982	39	0.33	0.11-0.92	10	0
	YP	CM	1978, 82	4	0.32	0.28-0.40	0	0
		LS	1981, 1982	58	0.32	0.15-0.75	7	0
	WF	LS	1981	43	0.06	0.02-0.13	0	0
Aswapiswaner 63/13 E	YP	CM	1971, 1982	4	0.27	0.14-0.44	0	0
		LS	1982	5	0.35	0.25-0.40	0	0
Athridge 101-102/19-20 W	NP	CM	1974	1	0.25	0.25-0.25	0	0
	YP	CM	1974	1	0.32	0.32-0.32	0	0
Atkinson 80-81/17-18 E	WF	CM	1970	1	0.28	0.28-0.28	0	0
	NP	CM	1970, 71, 80, 81	5	0.26	0.20-0.31	0	0
		LS	1980, 1981	15	0.24	0.11-0.38	0	0
	YP	CM	1970, 1979 1981	4	0.26	0.19-0.36	0	0
		LS	1979, 1981	17	0.28	0.12-0.51	0	0
Baldock 86-87/3-4 W	NP	CM	1972, 1979 1980, 1982	8	0.39	0.17-0.62	13	0
		LS	1978-1982	108	0.34	0.08-0.79	8	0
	YP	CM	1970, 1972 1979, 1982	9	0.33	0.18-0.43	0	0
		LS	1978-1982	104	0.34	0.12-0.66	13	0
	WF	CM	1970	1	0.12	0.12-0.12	0	0
		LS	1978, 1981	16	0.04	0.01-0.09	0	0
Barlow 91-92/13 W	NP	CM	1978, 1979	6	0.73	0.42-1.18	67	17
		LS	1978, 1979	30	0.60	0.12-1.69	43	20
	YP	CM	1978, 1979 1982	9	0.41	0.31-0.51	0	0
		LS	1978, 1982	54	0.45	0.20-0.78	26	0

LOCATION	SPECIES	SURVEY TYPE	YEARS	N	CONCENTRATION (ug/g)		% GREATER THAN	
					MEAN	RANGE	0.54	1.00
Barrington 91-92/18 W	NP	CM	1970, 71 1974-78, 83	18	0.53	0.33-0.75	22	0
		LS	1978, 83	10	0.62	0.30-1.06	70	10
	YP	CM	1970, 71, 74-78, 82, 83	11	0.35	0.13-0.47	0	0
		LS	1978, 82, 83	15	0.39	0.25-0.55	7	0
	WF	LS	1970	1	0.07	0.07-0.07	0	0
Batty 71/21-22 W	NP	CM	1972, 76, 77 81, 82	7	0.43	0.21-0.87	29	0
		LS	1981, 1982	16	0.55	0.16-1.25	50	13
	YP	CM	1971, 72, 76 77, 81, 82	16	0.29	0.15-0.58	6	0
		LS	1981-82	17	0.36	0.08-0.94	24	0
Bear 70-71/10 E	NP	CM	1971, 1977-1979	14	0.24	0.14-0.36	0	0
		LS	1978-1979	15	0.18	0.07-0.34	0	0
	YP	CM	1971, 1977-1979	9	0.17	0.10-0.22	0	0
		LS	1978-1979	12	0.23	0.13-0.38	0	0
Bearhead 75-76/9E	NP	CM	1972, 1981	6	0.64	0.44-0.89	83	0
		LS	1978, 1981	13	0.44	0.18-0.95	23	0
	YP	CM	1972	2	0.40	0.27-0.53	0	0
Beaucage 86/20-21 W	NP	CM	1972, 1981	6	1.10	0.74-1.52	100	50
		LS	1979, 1982	31	0.44	0.43-1.70	94	65
	YP	CM	1979, 1982	6	1.17	0.93-1.40	100	83
		LS	1979, 1982	30	1.28	0.35-1.93	93	73
Beaverhill 61/18 E	NP	CM	1971, 77, 78 79, 81	21	0.27	0.10-0.50	0	0
		LS	1981	5	0.23	0.19-0.30	0	0
	YP	CM	1971, 77, 78 79, 81	18	0.15	0.09-0.21	0	0
		LS	1981	6	0.12	0.10-0.13	0	0

LOCATION	SPECIES	SURVEY TYPE	YEARS	N	CONCENTRATION (ug/g)		% GREATER THAN	
					MEAN	RANGE	0.54	1.00
Big Sand 102/14 W	NP	CM	1978-1980	15	0.55	0.26-0.83	53	0
		LS	1978-1980	50	0.50	0.14-1.17	30	2
	YP	CM	1977-1980	13	0.52	0.30-0.64	38	0
		LS	1978-1980	46	0.54	0.19-1.04	37	2
Bigstone 73/12-13 E	NP	CM	1981, 1982	2	0.38	0.33-0.44	0	0
		LS	1981, 1982	9	0.32	0.04-0.59	11	0
	YP	CM	1971, 81, 82	3	0.21	0.12-0.27	0	0
		LS	1981, 82	10	0.20	0.06-0.48	0	0
	WF	CM	1971		0.08	0.08-0.08	0	0
Billard 94/9 E	WF	CM	1970	1	0.08	0.08-0.08	0	0
	YP	CM	1970	1	0.33	0.33-0.33	0	0
Billy 94/17 W	NP	CM	1981	4	0.60	0.35-0.78	75	0
		LS	1981	14	0.69	0.33-1.12	64	14
	YP	CM	1981	4	0.37	0.27-0.46	0	0
		LS	1981	18	0.50	0.25-0.79	39	0
Birch	YP	CM	1971, 75	2	0.56	0.43-0.69	50	0
Bird 97/10 W	NP	CM	1979, 83	6	0.53	0.33-0.98	33	0
		LS	1979, 83	30	0.50	0.23-1.39	27	7
	YP	CM	1979	1	0.76	0.76-0.76	100	0
		LS	1979	6	0.88	0.52-1.00	83	0
Bissett 89/7 E	NP	CM	1971	1	0.27	0.27-0.27	0	0
Black 70/4 E	NP	CM	1972, 73	15	0.57	0.20-1.07	60	7
	YP	CM	1972, 73	6	0.33	0.27-0.40	0	0
Boisvert 93/20 W	NP	CM	1980	1	0.24	0.24-0.24	0	0
		LS	1980	7	0.23	0.14-0.31	0	0

LOCATION	SPECIES	SURVEY TYPE	YEARS	N	CONCENTRATION (ug/g) MEAN	RANGE	% GREATER THAN 0.54	1.00
Bolton 61/12 E	NP	CM	1974,76-1979	15	0.28	0.16-0.49	0	0
		LS	1979	5	0.20	0.12-0.27	0	0
	YP	CM	1974,76-1979	15	0.26	0.20-0.34	0	0
		LS	1979	5	0.30	0.15-0.60	20	0
Bowder 68/8 W	NP	CM	1976,81	2	0.41	0.27-0.55	50	0
		LS	1981	5	0.29	0.16-0.50	0	0
Brannigan 79-80/4 E	NP	CM	1981-83	5	0.43	0.15-0.77	20	0
		LS	1981-83	25	0.43	0.09-1.09	24	8
	YP	CM	1981-83	5	0.36	0.25-0.56	20	0
		LS	1981-83	29	0.40	0.16-0.97	21	0
Brisebois 97-98/20-21 W	NP	CM	1980	1	0.31	0.31-0.31	0	0
		LS	1980	10	0.30	0.19-0.55	10	0
	YP	LS	1980	5	0.34	0.24-0.40	0	0
Britton 80/28-29 W	NP	CM	1978,83	2	0.44	0.43-0.46	0	0
		LS	1983	5	0.43	0.25-0.61	20	0
	YP	CM	1978,80,83	3	0.43	0.20-0.56	33	0
		LS	1980,1983	10	0.59	0.38-0.86	60	0
Bruneau 69-70/3-4 W	NP	CM	1973-79,82	12	0.49	0.23-1.50	17	8
		LS	1978,79,82	20	0.39	0.10-0.75	10	0
	YP	CM	1973-1979	12	0.54	0.26-0.82	33	0
		LS	1978,1979	11	0.50	0.22-1.33	36	9
Brunne part 66/25 W	NP	CM	1973,77	3	0.16	0.14-0.17	0	0
	YP	CM	1973,77	4	0.21	0.08-0.26	0	0
Buckingham 78/2 E	NP	CM	1979	1	0.85	0.85-0.85	100	0
		LS	1979	5	0.86	0.51-1.13	80	40
	YP	CM	1979,81	2	0.36	0.29-0.43	0	0
		LS	1979,81	13	0.39	0.13-0.85	15	0

LOCATION	SPECIES	SURVEY TYPE	YEARS	N	CONCENTRATION (ug/g)		% GREATER THAN	
					MEAN	RANGE	0.54	1.00
Buckland 99-100/4-5 E	WF	CM	1970	1	0.07	0.07-0.07	0	0
Budgroup 75/27 W	YP	CM	1971	2	0.69	0.64-0.74	100	0
Bully 70-71/3 E	NP	CM	1977	1	0.35	0.35-0.35	0	0
	YP	CM	1977	1	0.27	0.27-0.27	0	0
Burntwood 74/18-19 W	NP	CM	1971,72 76,77,79,81	15	0.42	0.21-0.61	13	0
		LS	1979,81	16	0.44	0.15-0.82	31	0
	YP	CM	1971-74 76,77,79,81	22	0.28	0.13-0.37	0	0
		LS	1979,81	19	0.35	0.23-0.57	5	0
Burntwood River 82/6 E	NP	CM	1980	1	1.16	1.16-1.16	100	100
		LS	1980	12	1.11	0.79-1.56	100	58
	YP	CM	1980	1	1.41	1.41-1.41	100	100
		LS	1980	9	1.45	1.21-1.78	100	100
Butterfly 63/2 E	NP	CM	1976,78,80	8	0.26	0.15-0.52	0	0
		LS	1978,1980	20	0.30	0.08-0.68	10	0
	YP	CM	1975-78 1980,81	14	0.32	0.12-0.47	0	0
		LS	1978,80,81	23	0.22	0.13-0.50	0	0
Butnau 85/14 E	NP	LS	1982	7	0.32	0.13-0.50	0	0
	YP	LS	1982	20	0.43	0.20-0.87	25	0
	WF	LS	1982	2	0.09	0.07-0.12	0	0
Cacholotte 69/28 W	NP	CM	1976	1	0.31	0.31-0.31	0	0
	YP	CM	1976	1	0.32	0.32-0.32	0	0
Caldwell 88-89/6 E	YP	CM	1971,72	2	0.37	0.33-0.41	0	0

LOCATION	SPECIES	SURVEY TYPE	YEARS	N	CONCENTRATION (ug/g)		% GREATER THAN	
					MEAN	RANGE	0.54	1.00
Campbell 87/2 E	WF	CM	1971	1	0.11	0.11-0.11	0	0
	YP	CM	1971	1	0.29	0.29-0.29	0	0
Carlson Block 97/25 W	NP	CM	1982	1	0.69	0.69-0.69	100	0
		LS	1982	5	0.65	0.21-0.87	80	0
	YP	CM	1982	1	0.49	0.49-0.49	0	0
		LS	1982	5	0.50	0.39-0.59	20	0
Carriere 97/27 W	NP	CM	1973,80,81	3	0.42	0.36-0.48	0	0
		LS	1980,81	10	0.46	0.26-0.65	30	0
	YP	CM	1973,80,81	3	0.51	0.42-0.62	33	0
		LS	1980,81	10	0.63	0.41-0.94	60	0
Carrot River 67/10 E	NP	CM	1972,75,82	10	0.44	0.33-0.54	0	0
		LS	1982	5	0.36	0.20-0.52	0	0
	YP	CM	1972,75,77,82	9	0.39	0.26-0.53	0	0
		LS	1982	5	0.38	0.23-0.49	0	0
Cauchon 74/6 E	WF	CM	1970,73	5	0.11	0.06-0.16	0	0
	NP	CM	1970-1983	101	0.58	0.10-1.14	56	3
		LS	1971,78,79 80,83	100	0.65	0.24-1.48	60	9
	YP	CM	1970-76,78	106	0.46	0.18-0.89	78	0
		LS	1971,78,80	105	0.55	0.23-1.63	42	3
Chapman 92/5 W	NP	CM	1979,81,82	4	0.33	0.31-0.34	0	0
		LS	1979,81,82	20	0.36	0.18-0.91	20	0
	YP	CM	1979,81,82	4	0.41	0.37-0.46	0	0
		LS	1979,81,82	23	0.45	0.17-0.98	30	0
Chatwin Block 114/20 W	YP	CM	1980,81	4	0.75	0.65-0.85	100	0
		LS	1980,81	19	0.81	0.42-1.24	89	21
Chepil 90-91/20 W	NP	CM	1981	2	0.66	0.48-0.84	50	0
		LS	1981	10	0.65	0.23-1.05	60	10
	YP	CM	1981	2	0.37	0.35-0.39	0	0
		LS	1981	12	0.41	0.23-0.65	25	0

LOCATION	SPECIES	SURVEY TYPE	YEARS	N	CONCENTRATION (ug/g)		% GREATER THAN	
					MEAN	RANGE	0.54	1.00
Chicken 87/21 W	WF	CM	1970	1	0.06	0.06-0.06	0	0
Chipewyan 103-104/6-7 W	NP	CM	1982	1	0.62	0.62-0.62	100	0
		LS	1982	5	0.56	0.35-0.89	100	0
	YP	CM	1982, 1983	3	0.64	0.38-0.83	67	0
		LS	1982, 1983	15	0.68	0.23-1.43	60	20
Christie 91/4 E	YP	CM	1971	3	0.28	0.22-0.35	0	0
Chuipka 96/18 W	YP	CM	1981, 83	5	0.47	0.37-0.66	20	0
		LS	1981, 83	29	0.52	0.29-1.42	24	3
Churchill 111/20 E	WF	CM	1970	1	0.04	0.04-0.04	0	0
	NP	CM	1971, 73 75, 81-83,	12	0.45	0.12-0.89	25	0
		LS	1981-83	15	0.32	0.09-0.84	7	0
	YP	CM	1970-76, 1981-83	29	0.26	0.09-0.45	0	0
		LS	1981-83	29	0.22	0.08-0.60	3	0
Clifton 112/11 W	NP	CM	1982	2	0.48	0.46-0.51	0	0
		LS	1982	10	0.50	0.24-0.67	50	0
	YP	CM	1982	2	0.64	0.61-0.67	100	0
		LS	1982	10	0.72	0.38-0.97	70	0
Cockeram 90/22 W	NP	CM	1971	1	0.26	0.26-0.26	0	0
	YP	CM	1971	1	0.44	0.44-0.44	0	0
Colen 64/14-15 E	WF	CM	1970	1	0.12	0.12-0.12	0	0
	NP	CM	1970, 1971	7	0.21	0.17-0.27	0	0
	YP	CM	1970, 1971	7	0.17	0.07-0.26	0	0
Cook 67/25 W	YP	CM	1981	1	0.42	0.42-0.42	0	0
		LS	1981	6	0.45	0.22-0.78	17	0

LOCATION	SPECIES	SURVEY TYPE	YEARS	N	CONCENTRATION (ug/g)		% GREATER THAN	
					MEAN	RANGE	0.54	1.00
Cormorant 60-61/22-23 W	WF	CM	1970,73	2	0.17	0.05-0.30	0	0
	NP	CM	1971-74,76 78,81,83	45	0.33	0.10-0.60	4	0
		LS	1971,81,83	96	0.33	0.15-0.73	4	0
	YP	CM	1971-74, 1976-78 80,81,83	43	0.30	0.15-0.59	2	0
		LS	1971,80, 81,83	110	0.30	0.13-0.80	6	0
Costello 83/17-18 W	NP	CM	1977	1	0.83	0.83-0.83	100	0
	YP	CM	1977,79	2	0.38	0.30-0.47	0	0
		LS	1979	6	0.51	0.21-1.51	17	17
Cotton (Camping) 70/5 E	NP	CM	1970,71,73, 74,76-81	22	0.24	0.10-0.37	0	0
		LS	1978-81	31	0.28	0.11-0.57	3	0
	YP	CM	1978-81	24	0.22	0.10-0.33	0	0
		LS	1978-81	30	0.26	0.12-0.49	0	0
Cousins 90/7 W	WF	LS	1975	25	0.13	0.05-0.23	0	0
	NP	CM	1977	1	0.51	0.51-0.51	0	0
Craig 88-89/27-28 W	NP	CM	1980	2	0.59	0.56-0.62	100	0
		LS	1980	11	0.61	0.33-0.99	55	0
	YP	CM	1980	2	0.71	0.68-0.75	100	0
		LS	1980	12	0.86	0.46-1.55	83	25
Cranberry 64-65/26 W	WF	CM	1970	1	0.07	0.07-0.07	0	0
	NP	CM	1970	1	0.14	0.14-0.14	0	0
	YP	CM	1970	1	0.20	0.20-0.20	0	0

LOCATION	SPECIES	SURVEY TYPE	YEARS	N	CONCENTRATION (ug/g)		% GREATER THAN	
					MEAN	RANGE	0.54	1.00
Cross (CFL The Pas) 66/1 W	WF	CM	1970	6	0.09	0.05-0.17	0	0
	NP	CM	1970,81, 77-79,82	14	0.42	0.19-0.76	7	0
		LS	1971,78,79,82	121	0.38	0.12-1.27	17	1
	YP	CM	1970-73 1975-79,82	38	0.35	0.17-0.64	3	0
		LS	1971,78,79 1982	115	0.34	0.05-0.88	11	0
Crying 86/9-10 E	NP	LS	1982	25	0.98	0.45-1.98	84	48
	YP	LS	1982	13	0.85	0.41-1.08	92	23
Cubs (Wilkins) 71/7 E	NP	CM	1972,80,82	7	0.36	0.22-0.61	14	0
		LS	1980,82	15	0.52	0.22-0.91	33	0
	YP	CM	1972,80,82	6	0.42	0.27-0.72	17	0
		LS	1980,82	17	0.63	0.18-1.36	41	18
Cuddle 74/11-12 E	NP	CM	1972,78,82	4	0.35	0.26-0.42	0	0
		LS	1978,82	34	0.29	0.05-0.72	12	0
	YP	CM	1972,78,82	3	0.25	0.19-0.30	0	0
		LS	1978,82	32	0.20	0.06-0.50	0	0
Cygnet Lake Group 89-90/16-17 E	NP	LS	1982	27	0.49	0.14-1.35	37	4
	YP	LS	1982	2	0.51	0.50-0.52	0	0
Dafoe 77-78/8 E	NP	CM	1970-72 1976-78	18	0.34	0.14-0.70	6	0
		LS	1978	5	0.47	0.24-0.68	40	0
	YP	CM	1970,72,74 76-78,80	13	0.34	0.25-0.42	0	0
		LS	1978,80	12	0.33	0.16-0.53	0	0
Denison 98/13 W	NP	CM	1977,1983	6	0.41	0.28-0.53	0	0
		LS	1983	20	0.36	0.18-0.71	10	0
	YP	CM	1977,83	6	0.46	0.34-0.55	17	0
		LS	1983	16	0.45	0.23-0.76	19	0

LOCATION	SPECIES	SURVEY TYPE	YEARS	N	CONCENTRATION (ug/g)		% GREATER THAN	
					MEAN	RANGE	0.54	1.00
Dobbyn 86/22-23 W	NP	CM	1982	1	0.44	0.44-0.44	0	0
		LS	1982	5	0.47	0.36-0.55	40	0
	YP	CM	1977,82	2	0.69	0.62-0.77	100	0
		LS	1982	5	0.83	0.52-1.02	80	20
Dolomite 63/21 W	WF	CM	1973	1	0.04	0.04-0.04	0	0
	NP	CM	1972-74,77,81	12	0.28	0.19-0.40	0	0
		LS	1981	5	0.41	0.30-0.57	20	0
	YP	CM	1972-74,77	10	0.25	0.19-0.38	0	0
Dow (see Muddy) 69/21 W	NP	LS	1971	47	0.58	0.06-1.10	51	6
	YP	LS	1971	96	0.56	0.21-1.17	48	3
Drunken 64/7 W	NP	CM	1970-1983	47	0.61	0.17-1.67	51	11
		LS	1972,1975 1978,1979 1981,1983	118	0.71	0.11-2.58	54	19
	YP	CM	1970-1983	43	0.65	0.22-1.15	65	5
		LS	1972,1975 1976,78,79 1981-1983	240	0.61	0.15-1.70	49	12
	WF	CM	1970	1	0.16	0.16-0.16	0	0
Duck (CFL) 67-68/5 W	WF	CM	1970	1	0.06	0.06-0.06	0	0
	NP	CM	1970-72 75,76	44	0.63	0.25-1.23	55	14
		LS	1971	63	0.56	0.25-1.29	49	6
	YP	CM	1970-72 75,76	43	0.44	0.13-0.76	19	0
		LS	1971	98	0.67	0.27-1.13	69	8
Dugas 70/4 W	NP	CM	1973,71,75 76,78,80	11	0.23	0.11-0.42	0	0
		LS	1978,80	10	0.42	0.20-0.95	20	0
	YP	CM	1971,73,75 76,78	10	0.20	0.12-0.59	10	0
		LS	1978	6	0.22	0.13-0.26	0	0

LOCATION	SPECIES	SURVEY TYPE	YEARS	N	CONCENTRATION (ug/g)		% GREATER THAN	
					MEAN	RANGE	0.54	1.00
Dunphy 88-89/26 W	WF	CM	1970	1	0.02	0.02-0.02	0	0
	NP	CM	1970,71,73	8	0.23	0.14-0.33	0	0
	YP	CM	1970,71,73	6	0.25	0.21-0.28	0	0
Dunsheath 93/21 W	NP	CM	1978,79	3	0.55	0.43-0.72	33	0
		LS	1978,1980	16	0.43	0.25-0.80	13	0
	YP	CM	1978,79,82	4	0.42	0.32-0.59	25	0
		LS	1978,79,82	17	0.44	0.26-0.92	12	0
Dyce 62/18 W	NP	CM	1971,73,77 79,81	9	0.38	0.34-0.44	0	0
		LS	1979,81	10	0.36	0.21-0.80	10	0
	YP	CM	1971,77,79,81	7	0.48	0.35-0.55	14	0
		LS	1979,81	16	0.45	0.29-0.76	25	0
Dysart 102/1 W	NP	CM	1982	1	0.69	0.69-0.69	100	0
		LS	1982	5	0.61	0.33-1.32	40	20
	YP	CM	1982	1	0.56	0.56-0.56	100	0
		LS	1982	5	0.65	0.47-0.97	60	0
Eden 88/18 W	NP	CM	1972,73,76	13	0.43	0.30-0.57	8	0
	YP	CM	1977,73,76	6	0.35	0.27-0.43	0	0
Edmund 66/5 E	NP	CM	1971,76-80	16	0.23	0.16-0.32	0	0
	YP	CM	1971,76-80	17	0.16	0.13-0.24	0	0
Egg 62/27 W	NP	CM	1976-78,80	8	0.27	0.15-0.37	0	0
		LS	1978,80	11	0.20	0.09-0.39	0	0
	YP	CM	1976-78,80	6	0.34	0.29-0.45	0	0
		LS	1978,80	11	0.40	0.28-0.63	18	0
Election 64-65/25 W	WF	CM	1970	1	0.08	0.08-0.08	0	0
	NP	CM	1970,71,77	5	0.54	0.42-0.64	40	0
		LS	1971,72	91	0.46	0.21-1.00	26	0
	YP	CM	1977	1	0.34	0.34-0.34	0	0

LOCATION	SPECIES	SURVEY TYPE	YEARS	N	CONCENTRATION (ug/g)		% GREATER THAN	
					MEAN	RANGE	0.54	1.00
Enatik 93/14 W	NP	CM	1977,80-82	8	0.52	0.27-0.77	38	0
		LS	1980-82	38	0.48	0.16-1.21	32	8
	YP	CM	1977,80-82	10	0.49	0.23-1.03	30	10
		LS	1980-82	40	0.54	0.15-1.78	38	5
End 92/19 W	NP	CM	1980	2	0.78	0.57-0.99	100	0
		LS	1980	11	0.70	0.45-1.60	73	9
	YP	CM	1980	2	0.61	0.61-0.62	100	0
		LS	1980	10	0.70	0.40-1.26	50	20
Enger Block 106/25-26 W	YP	CM	1979	2	0.69	0.65-0.73	100	0
		LS	1979	12	0.73	0.28-1.15	92	17
Eyrie Block 99/21 W	NP	CM	1978-80,82	5	0.37	0.26-0.49	0	0
		LS	1978-80,82	26	0.34	0.13-0.76	8	0
	YP	CM	1978-80,82	4	0.31	0.27-0.37	0	0
		LS	1978-80,82	28	0.33	0.17-0.63	4	0
Fagnan 107/24 W	NP	CM	1982	1	0.69	0.69-0.69	100	0
		LS	1982	5	0.52	0.22-1.20	20	20
	YP	CM	1982	2	0.64	0.53-0.75	50	0
		LS	1982	10	0.70	0.39-1.22	50	20
Fairy Rock 58/14-15 E	NP	CM	1978	1	0.57	0.57-0.57	100	0
		LS	1978	21	0.53	0.13-1.55	29	10
	YP	CM	1978	1	0.51	0.51-0.51	0	0
		LS	1978	19	0.45	0.12-0.91	37	0
Fay 69/24-25 W	NP	CM	1978	2	0.47	0.32-0.63	50	0
	YP	CM	1978	2	0.59	0.57-0.61	100	0
Fidler 94-95/4 E	WF	CM	1970	1	0.10	0.10-0.10	0	0

LOCATION	SPECIES	SURVEY TYPE	YEARS	N	CONCENTRATION (ug/g)		% GREATER THAN	
					MEAN	RANGE	0.54	1.00
Finch 87/22 W	NP	CM	1980,82,83	3	0.49	0.37-0.65	33	0
		LS	1980,82,83	15	0.47	0.20-1.01	20	7
	YP	CM	1980,82,83	3	0.45	0.35-0.57	33	0
		LS	1980,82,83	17	0.51	0.29-0.85	35	0
Fish 69/9 W	NP	CM	1971,74,75 77-79,81,83	12	0.37	0.14-0.64	17	0
		LS	1978,79,81,83	38	0.39	0.15-1.00	16	0
	YP	CM	1971,74,75 77-79,81,83	11	0.25	0.12-0.37	0	0
		LS	1978,79,81,83	51	0.26	0.11-0.60	2	0
Five Mile 70-71/8 W	NP	CM	1979	2	0.96	0.87-1.05	100	50
		LS	1979	10	0.96	0.60-1.28	100	50
	YP	CM	1979	2	0.62	0.47-0.78	50	0
		LS	1979	10	0.58	0.25-1.21	40	20
Footprint 78/10 W	NP	LS	1978,80,81,83	99	0.99	0.28-3.37	85	41
	YP	LS	1978,80,81,83	115	0.99	0.29-2.61	88	43
	WF	LS	1980,81,83	55	0.19	0.06-0.48	0	0
Fraser 91/14-15 W	NP	CM	1982	1	0.45	0.45-0.45	0	0
		LS	1982	5	0.40	0.33-0.54	0	2
	YP	CM	1982,83	3	0.30	0.21-0.42	0	0
		LS	1982,83	17	0.29	0.17-0.73	12	0
Gauer 92/3 W	NP	CM	1970,78,80-82	6	0.36	0.19-0.51	0	0
		LS	1978,80-82	49	0.28	0.09-0.77	6	0
	YP	CM	1970-72,78 1980-82	10	0.23	0.07-0.36	0	0
		LS	1978,80-82	48	0.23	0.10-0.53	0	0
	WF	CM	1970	1	0.13	0.13-0.13	0	0
		LS	1981	33	0.04	0.01-0.12	0	0
Gestur 74/28-29 W	NP	CM	1972	1	0.39	0.39-0.39	0	0
	YP	CM	1972	2	0.42	0.26-0.59	50	0

LOCATION	SPECIES	SURVEY TYPE	YEARS	N	CONCENTRATION (ug/g)		% GREATER THAN	
					MEAN	RANGE	0.54	1.00
Girouard 75/26 W	NP	CM	1971-73	12	0.72	0.40-1.10	83	8
	YP	CM	1971-73	12	0.69	0.37-1.18	67	8
Gods 66/22 E	WF	CM	1972-73	3	0.04	0.03-0.05	0	0
	NP	CM	1972,73,76	14	0.20	0.10-0.34	0	0
	YP	CM	1971-73,76 78,81	16	0.24	0.15-0.41	0	0
		LS	1981	5	0.26	0.19-0.39	0	0
Goldsand 92/23 W	WF	CM	1970	1	0.11	0.11-0.11	0	0
	NP	CM	1970,71,73	7	0.22	0.12-0.44	0	0
		LS	1983	5	0.36	0.15-0.85	20	0
	YP	CM	1970,71	7	0.23	0.17-0.36	0	0
Goose 63/27-28 W	NP	CM	1972,74 77,78,81	7	0.34	0.07-0.57	14	0
		LS	1981	10	0.48	0.29-0.69	30	0
	YP	CM	1972,77,78,81	7	0.37	0.28-0.48	0	0
		LS	1981	6	0.55	0.38-0.73	50	0
Goose 61/20 E	WF	CM	1970	3	0.15	0.07-0.29	0	0
	NP	CM	1970-73,76,80	19	0.40	0.18-0.85	11	0
	YP	CM	1970-72,80	14	0.32	0.23-0.45	0	0
Goulet 74/8 E	NP	CM	1972,76,78 79,81,82	12	0.50	0.23-1.20	33	17
		LS	78,79,81,82	47	0.57	0.19-1.70	43	17
	YP	CM	1976-82	11	0.37	0.21-0.50	0	0
		LS	1978-82	52	0.38	0.14-1.14	13	2
Granville 84/19 W	NP	CM	1971,73,74 76-80,82	14	0.45	0.14-0.65	36	0
		LS	1976,78-82	44	0.46	0.05-0.74	34	0
	YP	CM	1970-1978 1980-82	20	0.26	0.10-0.41	0	0
		LS	1976,78,80-82	105	0.26	0.07-0.65	3	0
	WF	CM	1970	1	0.07	0.07-0.07	0	0
		LS	1976	1	0.05	0.05-0.05	0	0

LOCATION	SPECIES	SURVEY TYPE	YEARS	N	CONCENTRATION (ug/g)		% GREATER THAN	
					MEAN	RANGE	0.54	1.00
Gull 84-85/13-14 E	NP	LS	1982	24	1.29	0.72-1.76	100	88
	YP	LS	1982	25	0.90	0.56-1.31	100	32
Gunisao 52-53/9 E	NP	LS	1981	2	0.15	0.11-0.20	0	0
	YP	LS	1981	30	0.13	0.05-0.33	0	0
	WF	LS	1981	9	0.04	0.02-0.08	0	0
Guthrie 72-73/21 W	NP	CM	1973-81	8	0.20	0.11-0.41	0	0
		LS	1978-79, 81	15	0.24	0.08-0.77	7	0
	YP	CM	1973-79, 81	8	0.16	0.09-0.21	0	0
		LS	1978, 79, 81	18	0.13	0.06-0.30	0	0
Halfway 70/7 W	WF	CM	1970, 73	2	0.04	0.02-0.07	0	0
	NP	CM	1970-73 75, 76, 81	17	0.18	0.05-0.58	6	0
		LS	1981	8	0.15	0.09-0.23	0	0
Harding 83/7-6 W	WF	LS	1982	1	0.02	0.02-0.02	0	0
	NP	LS	1982	5	0.67	0.48-0.84	80	0
	YP	LS	1982	17	0.29	0.14-0.52	0	0
Herb (Wekusko) 66-67/16-17 W	WF	CM	1970, 73	3	0.05	0.02-0.10	0	0
	NP	CM	1970-76, 81	18	0.20	0.09-0.40	0	0
		LS	1981	5	0.33	0.19-0.56	20	0
	YP	CM	1970-76, 81	18	0.15	0.08-0.44	0	0
		LS	1981	6	0.10	0.07-0.16	0	0
Herblet (Little Herb) 68-69/16 W	NP	CM	1971, 73 74, 76, 78, 79	7	0.32	0.17-0.45	0	0
		LS	78, 79	10	0.37	0.19-0.62	20	0
Herman 92/19 W	NP	CM	1974, 79, 81	3	0.60	0.56-0.66	100	0
		LS	1979, 81	11	0.59	0.24-1.43	55	9
	YP	CM	1974, 79, 80, 81	4	0.70	0.43-0.95	75	0
		LS	79, 80, 81	16	0.81	0.31-1.32	75	25

LOCATION	SPECIES	SURVEY TYPE	YEARS	N	CONCENTRATION (ug/g)		% GREATER THAN	
					MEAN	RANGE	0.54	1.00
High Hill 76/12 E	WF	CM	1970	2	0.11	0.07-0.15	0	0
	NP	CM	1970,71	2	0.26	0.23-0.29	0	0
	YP	CM	1970,71,78	3	0.23	0.20-0.27	0	0
		LS	1978	5	0.25	0.15-0.40	0	0
High Rock 78-79/20 W	NP	CM	1972,77,78 81,82	7	0.46	0.23-0.72	29	0
		LS	1978,81,82	16	0.47	0.15-1.22	31	6
	YP	CM	1972,77,78 81,82	15	0.32	0.25-0.44	0	0
		LS	1978,81,82	30	0.37	0.13-0.87	10	0
Hill 62/9-10 W	NP	CM	1971,77	5	1.11	0.54-1.72	80	60
	YP	CM	1977	4	1.16	0.93-1.46	100	50
Holiday 96/21 W	NP	CM	1980	1	0.61	0.61-0.61	100	0
		LS	1980	5	0.66	0.47-0.82	80	0
	YP	CM	1980	1	0.78	0.78-0.78	100	0
		LS	1980	5	0.91	0.72-1.12	100	20
Holmes 93/5 E	NP	CM	1982	1	0.21	0.21-0.21	0	0
		LS	1982	13	0.26	0.14-0.45	0	0
	YP	CM	1971	3	0.14	0.12-0.16	0	0
		LS	1982	2	0.05	0.03-0.07	0	0
Hook 108/25 W	NP	CM	1972	1	0.88	0.88-0.88	100	0
	YP	CM	1972	2	0.67	0.63-0.72	100	0
Horseshoe 66/26 W	NP	CM	1978,1980	3	0.19	0.12-0.30	0	0
	YP	CM	1980	2	0.40	0.27-0.54	0	0
Horseshoe (Hjalmarson) 100/23 W	NP	CM	1976	3	0.24	0.15-0.43	0	0
		LS	1978,1980	15	0.20	0.08-0.53	0	0
	YP	CM	1976	4	0.29	0.27-0.32	0	0
		LS	1980	10	0.34	0.16-0.59	10	0

LOCATION	SPECIES	SURVEY TYPE	YEARS	N	CONCENTRATION (ug/g)		% GREATER THAN	
					MEAN	RANGE	0.54	1.00
Issett 87-88/12-13 W	WF	LS	1975,78,82,83	78	0.18	0.02-0.40	0	0
	NP	LS	1978,82,83	66	0.95	0.36-2.28	88	39
	YP	LS	1978,82,83	56	1.19	0.20-2.71	73	57
Jackpine 101/25 W	NP	CM	1971	1	0.15	0.15-0.15	0	0
James 95/15 W	NP	LS	1977	15	0.55	0.32-0.95	47	0
Joey 74/5 W	NP	CM	1981	2	0.24	0.23-0.26	0	0
		LS	1981	10	0.25	0.13-0.29	0	0
	YP	CM	1981	2	0.23	0.21-0.25	0	0
		LS	1981	10	0.23	0.11-0.63	10	0
Johnson 69/16 W	NP	CM	1981	1	0.36	0.36-0.36	0	0
		LS	1981	5	0.40	0.26-0.53	0	0
	YP	CM	1981	1	0.54	0.54-0.54	0	0
		LS	1981	5	0.56	0.33-0.74	60	0
Jordon 99/18 W	NP	CM	1978,82	4	0.82	0.76-0.93	100	0
		LS	1978,82	20	0.81	0.42-1.23	90	15
	YP	CM	1978,82	7	0.88	0.77-1.01	100	14
		LS	1978,82	36	0.90	0.27-1.36	81	47
Kadeniuk 86/25 W	NP	CM	1980	1	0.75	0.75-0.75	100	0
		LS	1980	5	0.59	0.18-1.84	20	20
	YP	CM	1980	1	0.21	0.21-0.21	0	0
		LS	1980	12	0.25	0.12-0.34	0	0
Kallie cahoolie 60-61/14 E	NP	CM	1977,78	3	0.21	0.18-0.23	0	0
	YP	CM	1977,78	4	0.31	0.24-0.36	0	0

LOCATION	SPECIES	SURVEY TYPE	YEARS	N	CONCENTRATION (ug/g)		% GREATER THAN	
					MEAN	RANGE	0.54	1.00
Kamak 99/19 W	NP	CM	1982	1	0.37	0.37-0.37	0	0
		LS	1982	10	0.30	0.22-0.50	0	0
	YP	CM	1982	1	0.62	0.62-0.62	100	0
		LS	1982	5	0.65	0.39-1.12	60	20
Kaminitiskayowk 105-106/24-25 W	NP	CM	1982	4	0.41	0.21-0.58	50	0
		LS	1982	10	0.40	0.24-0.81	10	0
	YP	CM	1982	3	0.52	0.40-0.68	33	0
		LS	1982	10	0.67	0.44-0.85	60	0
Kamuchawie 84-85/29 W	NP	CM	1979	1	0.96	0.96-0.96	100	0
		LS	1979	5	0.82	0.29-1.39	60	40
	YP	CM	1979	1	0.84	0.84-0.84	100	0
		LS	1979	5	0.93	0.57-1.31	100	40
Kipahigan (Barrier) 72-73/29 W	WF	CM	1970,72	2	0.06	0.05-0.08	0	0
	NP	CM	1970-74 1976-78,81	19	0.25	0.14-0.41	0	0
		LS	1978,81	11	0.26	0.06-0.46	0	0
	YP	CM	1970-74 1976-78,81	22	0.22	0.10-0.32	0	0
		LS	1978,81	12	0.22	0.10-0.37	0	0
Kiski 66-67/10 W	NP	CM	1971-73 1975-79,81	11	0.29	0.12-0.75	9	0
		LS	1978,79,81	16	0.68	0.13-3.99	44	6
	YP	CM	1971-73 1975-79,81	10	0.26	0.16-0.48	0	0
		LS	1978,79,81	19	0.22	0.12-0.42	0	0
Kiskitto 61/8 W	NP	CM	1970-73 1975-79,82	61	0.50	0.10-1.25	36	5
		LS	1971,78,79,82	124	0.37	0.10-1.45	15	3
	YP	CM	1970-76 1978,79,82	27	0.29	0.17-0.49	0	0
		LS	1971,78,79,82	129	0.37	0.09-0.91	8	0
	WF	CM	1970	3	0.09	0.07-0.31	0	0

LOCATION	SPECIES	SURVEY TYPE	YEARS	N	CONCENTRATION (ug/g)		% GREATER THAN	
					MEAN	RANGE	0.54	1.00
Kiskittogisu (Grassy) 60/6-7 W	NP	CM	1970,72-82	54	0.40	0.20-0.79	17	0
		LS	1971,78-82	122	0.51	0.19-1.33	33	5
	YP	CM	1970,1972, 1973,75-82	33	0.30	0.19-0.47	0	0
		LS	1971,78-82	140	0.33	0.12-0.80	4	0
	WF	CM	1970	3	0.07	0.04-0.09	0	0
Kissaynew 69/27-28 W	WF	CM	1970	1	0.04	0.04-0.04	0	0
	NP	CM	1970-72,74-78	17	0.33	0.10-0.49	0	0
		LS	1978	5	0.40	0.30-0.52	0	0
	YP	CM	1970,71,74-78	20	0.28	0.13-0.43	0	0
		LS	1978	6	0.38	0.09-0.57	17	0
Kississing 71/26 W	WF	CM	1970	1	0.02	0.02-0.02	0	0
	NP	CM	1970,71,74-78	8	0.19	0.13-0.30	0	0
		LS	1978	5	0.14	0.07-0.18	0	0
	YP	CM	1970,71,74-78	6	0.14	0.09-0.21	0	0
		LS	1978	6	0.12	0.07-0.17	0	0
Kistigan 65/10 E	WF	CM	1970	1	0.16	0.16-0.16	0	0
	NP	CM	1970,71,77	6	0.21	0.16-0.29	0	0
	YP	CM	1970,71,77	6	0.17	0.10-0.30	0	0
Knee 69/18-19 E	WF	CM	1970,71	2	0.04	0.03-0.06	0	0
	NP	CM	1970-72,76-80	26	0.25	0.12-0.39	0	0
	YP	CM	1970-72 76-80,82	17	0.15	0.09-0.23	0	0
		LS	1982	6	0.13	0.08-0.19	0	0
Kustra 102-103/17-18 W	NP	CM	1977,78	2	0.29	0.29-0.30	0	0
		LS	1978	5	0.31	0.25-0.37	0	0
	YP	CM	1977,78	5	0.45	0.35-0.56	20	0
		LS	1978	5	0.55	0.33-0.77	40	0

LOCATION	SPECIES	SURVEY TYPE	YEARS	N	CONCENTRATION (ug/g)		% GREATER THAN	
					MEAN	RANGE	0.54	1.00
Lac Brochet 110-111/25 W	WF	CM	1970	1	0.07	0.07-0.07	0	0
	NP	CM	1972	2	0.32	0.32-0.33	0	0
	YP	CM	1972, 77, 78	8	0.50	0.15-0.85	25	0
		LS	1978	5	0.54	0.49-0.61	40	0
Lafontaine 88/20 W	YP	CM	1982	1	0.82	0.82-0.82	100	0
		LS	1982	5	0.93	0.50-1.20	80	60
Landing 72-73/1-2 E	WF	CM	1971	1	0.02	0.02-0.02	0	0
	NP	CM	1971-76 78-79, 81-82	15	0.25	0.11-0.43	0	0
		LS	1978, 79, 81, 82	25	0.28	0.15-0.56	4	0
	YP	CM	1971-79, 82	16	0.20	0.11-0.32	0	0
		LS	78, 79, 82	18	0.20	0.14-0.31	0	0
Laurie 87-88/28-29 W	NP	CM	1972, 74, 79	19	1.06	0.69-1.54	100	53
		LS	1976, 79	67	1.12	0.75-2.25	100	61
	YP	CM	1972-74, 76, 79	11	0.84	0.70-1.19	100	9
		LS	1976, 79	64	0.85	0.43-1.62	92	23
Lawford 63-64/5-6 E	NP	CM	1978, 80, 81	4	0.34	0.27-0.49	0	0
		LS	1978, 80, 81	18	0.31	0.17-0.49	0	0
	YP	CM	1971, 76 78, 80, 81	7	0.31	0.25-0.38	0	0
		LS	78, 80, 81	17	0.33	0.19-0.68	6	0
Le Clair 97/18 W	NP	CM	1977, 80, 81, 83	4	0.88	0.61-1.30	100	25
		LS	1980, 81, 83	15	0.86	0.33-1.63	87	27
	YP	CM	1977, 79, 80-83	8	0.54	0.39-0.87	38	0
		LS	1979-83	43	0.63	0.27-1.13	56	7
Leftrook 81-82/8 W	NP	CM	1977, 81	2	0.29	0.25-0.33	0	0
		LS	1981	5	0.27	0.10-0.40	0	0
	YP	CM	1972, 76, 77 1981-83	7	0.33	0.14-1.01	14	14
		LS	1981-83	25	0.45	0.07-1.57	20	12

LOCATION	SPECIES	SURVEY TYPE	YEARS	N	CONCENTRATION (ug/g)		% GREATER THAN	
					MEAN	RANGE	0.54	1.00
Limestone 87/10-11 E	NP	CM	1972,76,79,81	7	0.51	0.29-0.77	43	0
		LS	1979,81	11	0.36	0.22-0.66	9	0
	YP	CM	1970-74,76 78,79,81	18	0.34	0.07-0.48	0	0
		LS	1978,79,81	19	0.39	0.19-0.83	11	0
Limestone Point 71/20 W	NP	CM	1976,77	4	0.81	0.62-1.18	100	25
	YP	CM	1976,77	8	0.47	0.32-0.62	13	0
Little Bolton 60/8-9 E	NP	CM	1978	1	0.66	0.66-0.66	100	0
		LS	1978	5	0.66	0.19-1.22	40	0
	YP	CM	1976-78	9	0.30	0.16-0.39	0	0
		LS	1978	14	0.25	0.13-0.63	7	0
Little Playgreen 58/3 W	WF	LS	1980	37	0.03	0.01-0.07	0	0
	NP	LS	1981	12	0.31	0.10-0.97	17	0
	YP	LS	1981	26	0.21	0.07-0.60	4	0
Logan 63/9 E	NP	CM	1978,81	4	0.27	0.11-0.34	0	0
		LS	1981	11	0.20	0.05-0.78	9	0
	YP	CM	1978,81,82	4	0.18	0.14-0.26	0	0
		LS	1981,82	19	0.17	0.08-0.28	0	0
Loon 79/29 W	NP	CM	1973	1	0.50	0.50-0.50	0	0
	YP	CM	1973,81	2	0.52	0.43-0.61	50	0
		LS	1981	6	0.62	0.21-1.42	33	17
Loon River 80/29 W	NP	CM	1981	1	0.53	0.53-0.53	0	0
		LS	1981	5	0.43	0.19-0.88	40	0
	YP	CM	1981	1	0.67	0.67-0.67	100	0
		LS	1981	6	0.70	0.41-1.11	67	17
MacBride 91/16 W	NP	CM	1977,81	2	0.45	0.23-0.67	50	0
		LS	1981	5	0.69	0.56-0.79	100	0
	YP	CM	1977,83	3	0.74	0.32-1.01	67	33
		LS	1983	6	0.94	0.66-1.24	100	50

LOCATION	SPECIES	SURVEY TYPE	YEARS	N	CONCENTRATION (ug/g)		% GREATER THAN	
					MEAN	RANGE	0.54	1.00
MacBride 90-91/16 W	YP	LS	1983	5	1.10	0.89-1.51	100	60
Macheewin 81/10 W	NP	CM	1981,82	3	0.62	0.33-0.98	33	0
		LS	1981,82	15	0.70	0.29-1.78	47	20
	YP	CM	1981,82	2	0.63	0.59-0.67	100	0
		LS	1981,82	12	0.71	0.33-0.96	75	0
Mackerracher 98/12 W	NP	CM	1977	5	0.31	0.25-0.35	0	0
	YP	CM	1972,77	5	0.37	0.28-0.48	0	0
Matheson Group 85/28 W	NP	CM	1979	1	0.92	0.92-0.92	100	0
		LS	1979	5	0.72	0.24-1.72	40	20
	YP	CM	1978,79	3	0.90	0.86-0.96	100	0
		LS	1978,79	15	0.96	0.69-1.38	100	40
Max Lake 64/10-11 E	NP	CM	1977,78,82	19	0.34	0.13-0.79	16	0
		LS	1978,82	32	0.19	0.04-0.74	3	0
	YP	CM	1977,78,81,82	16	0.33	0.23-0.46	0	0
		LS	1978,81,82	48	0.29	0.06-0.82	8	0
McCallum 82/28 W	NP	CM	1973	5	0.68	0.57-0.95	100	0
	YP	CM	1973	2	0.44	0.43-0.46	0	0
McCormick 71/2 E	NP	CM	1971,72,74	5	0.30	0.14-0.79	20	0
	YP	CM	1971,72,74	10	0.50	0.14-0.89	40	0
McGrath 117/20 W	NP	CM	1980,82	2	0.75	0.47-1.03	50	50
		LS	1980,82	11	0.64	0.33-1.93	55	9
	YP	CM	1979,80	3	0.58	0.43-0.84	33	0
		LS	1979,80	18	0.63	0.30-1.33	33	11
McKnight 81/24 W	NP	CM	1972,74,77,78	8	0.42	0.29-0.61	25	0
		LS	1978	5	0.46	0.16-0.62	40	0

LOCATION	SPECIES	SURVEY TYPE	YEARS	N	CONCENTRATION (ug/g)		% GREATER THAN	
					MEAN	RANGE	0.54	1.00
McLeod 95/8 W	NP	CM	1983	4	1.36	0.97-1.82	100	75
		LS	1983	19	1.22	0.51-2.44	95	63
	YP	CM	1983	1	0.23	0.23-0.23	0	0
		LS	1983	6	0.23	0.16-0.35	0	0
McPherson 96-97/12 W	NP	CM	1981,82	2	0.44	0.29-0.59	50	0
		LS	1977,81,82	21	0.42	0.07-0.96	19	0
	YP	CM	1981	1	0.29	0.29-0.29	0	0
		LS	1981	6	0.34	0.19-0.55	17	0
Melvin 94/18 W	NP	CM	1979-82	6	0.75	0.45-0.97	83	0
		LS	1979-82	31	0.75	0.14-1.48	74	23
	YP	CM	1979-82	6	0.80	0.57-1.10	100	17
		LS	1979-82	34	0.86	0.46-1.51	82	32
Midnight 75/6 E	YP	CM	1978	1	0.33	0.33-0.33	0	0
		LS	1978	6	0.40	0.18-0.69	17	0
Misty 114/25-26 W	NP	CM	1973,80,82	5	0.27	0.20-0.37	0	0
		LS	1980,82	15	0.30	0.14-0.55	7	0
	YP	CM	1973	2	0.32	0.31-0.34	0	0
Moak 80/1 W	NP	CM	1982,83	3	0.80	0.66-0.99	100	0
		LS	1982,83	16	0.79	0.40-1.75	88	19
	YP	CM	1983	1	0.70	0.70-0.70	100	0
		LS	1983	6	0.62	0.28-1.00	67	0
Molson 60/5 E	NP	CM	1970,77	3	0.24	0.14-0.35	0	0
		LS	1981	25	0.14	0.02-0.58	4	0
	YP	CM	1970,71,77,79	5	0.35	0.24-0.65	20	0
		LS	1979,81	11	0.42	0.05-0.94	27	0
	WF	CM	1970	1	0.08	0.08-0.08	0	0
		LS	1981	25	0.02	0.01-0.08	0	0
Moody 70/22 W	NP	CM	1971,77	2	0.60	0.53-0.68	50	0
	YP	CM	1971,77	2	0.28	0.20-0.37	0	0

LOCATION	SPECIES	SURVEY TYPE	YEARS	N	CONCENTRATION (ug/g)		% GREATER THAN	
					MEAN	RANGE	0.54	1.00
Moose 59/19 W	WF	CM	1970	1	0.05	0.05-0.05	0	0
	NP	CM	1970,71,73 74,76-78,80	62	0.32	0.12-0.71	8	0
		LS	1978,80	10	0.27	0.14-0.42	0	0
	YP	CM	1970,71, 73-78,80	37	0.19	0.08-0.49	0	0
		LS	1978,80	11	0.17	0.06-0.36	0	0
Moose (Ilford Coop) 86-87/14-15 E	NP	CM	1971	4	0.20	0.15-0.27	0	0
	YP	CM	1971	1	0.25	0.25-0.25	0	0
Moose (North Arm) 59/19 W	NP	CM	1972,75,76 77,79,80	54	0.38	0.13-0.90	13	0
		LS	1979,80	11	0.26	0.13-0.53	0	0
	YP	CM	1972,74-81	13	0.23	0.14-0.38	0	0
		LS	1978-81	24	0.20	0.09-0.47	0	0
Moosenose 81/12 E	NP	CM	1971,72	2	0.31	0.26-0.36	0	0
Morgan 66-67/18-19 W	WF	LS	1978	1	0.10	0.10-0.10	0	0
	NP	LS	1978	8	0.20	0.11-0.34	0	0
	YP	LS	1978	2	0.24	0.21-0.27	0	0
Morin 76/26 W	NP	CM	1971,81	7	0.68	0.49-0.82	86	0
		LS	1981	5	0.72	0.33-1.07	80	20
	YP	CM	1971-73,81	8	0.48	0.39-0.68	13	0
		LS	1981	6	0.70	0.33-0.97	67	0
Morton 67/20 W	WF	CM	1970	1	0.07	0.07-0.07	0	0
	NP	CM	1971	2	0.48	0.45-0.51	0	0
	YP	CM	1971	3	0.50	0.49-0.52	0	0

LOCATION	SPECIES	SURVEY TYPE	YEARS	N	CONCENTRATION (ug/g)		% GREATER THAN	
					MEAN	RANGE	0.54	1.00
Moss 99-100/8 W	NP	CM	1977,82,83	3	0.78	0.54-1.08	67	33
		LS	1982,83	10	1.02	0.27-1.78	90	40
	YP	CM	1977,82,83	4	0.65	0.27-1.28	50	25
		LS	1982,83	16	0.84	0.25-1.71	63	31
Mud 68/6 W	NP	CM	1972	3	0.80	0.29-1.16	67	33
	YP	CM	1972	5	0.52	0.44-0.65	40	0
Muddy (Dow) 69/21 W	NP	CM	1971	8	0.52	0.36-0.65	38	0
	YP	CM	1971,81	7	0.44	0.36-0.52	0	0
		LS	1981	6	0.38	0.18-0.78	17	0
Mulcahy 94/11 W	NP	CM	1981	1	0.50	0.50-0.50	0	0
		LS	1981	5	0.49	0.21-0.82	40	0
Mynarski East 82/11 W	NP	CM	1977,80-83	12	0.81	0.33-1.48	58	33
		LS	1978,80-83	146	0.64	0.13-2.95	42	19
	YP	CM	1977,80-83	18	0.67	0.33-1.71	50	11
		LS	1978,80-83	858	0.94	0.11-2.94	90	30
	WF	CM	1980	2	0.22	0.21-0.23	0	0
		LS	1980	17	0.24	0.14-0.41	0	0
Mynarski West 82/12 W	NP	CM	1982,83	3	0.57	0.48-0.65	67	0
		LS	1982,83	17	0.52	0.22-0.82	35	0
	YP	CM	1982,83	7	1.09	0.53-2.53	86	43
		LS	1982,83	47	1.05	0.24-2.99	77	32
Mystery 79/2 W	NP	CM	1982	6	1.09	0.58-1.44	100	50
		LS	1979,82,83	125	0.98	0.23-2.05	84	45
	YP	CM	1982	1	0.99	0.99-0.99	100	0
		LS	1979,82,83	60	1.02	0.13-1.75	87	52
	WF	LS	1983	21	0.19	0.05-0.45	0	0

LOCATION	SPECIES	SURVEY TYPE	YEARS	N	CONCENTRATION (ug/g)		% GREATER THAN	
					MEAN	RANGE	0.54	1.00
Naosap 67-68/26-27 W	WF	CM	1973	1	0.06	0.06-0.06	0	0
	NP	CM	1971-73	6	0.29	0.19-0.38	0	0
	YP	CM	1971-73	6	0.28	0.20-0.35	0	0
Narrows 71/24 W	YP	CM	1973	3	0.40	0.30-0.47	0	0
Nelson River (Mouth)	NP	CM	1975, 76, 78-83	36	0.96	0.52-1.47	97	36
		LS	1978-1983	137	0.95	0.18-2.39	88	37
	YP	CM	1975-1983	29	0.72	0.27-1.26	76	10
		LS	1979-1983	93	0.57	0.10-1.31	53	6
	WF	LS	1981	3	0.05	0.03-0.09	0	0
Nickel 113-114/14-15 W	NP	CM	1983	1	1.31	1.31-1.31	100	100
		LS	1983	5	1.30	0.78-1.61	100	80
	YP	CM	1983	1	0.43	0.43-0.43	0	0
		LS	1983	5	0.44	0.27-0.66	40	0
Northern Indian 96/1-2 E	NP	CM	1975, 76, 78 80, 82, 83	11	0.43	0.15-0.89	27	0
		LS	1978, 80, 82, 83	35	0.52	0.14-1.34	34	6
	YP	CM	1970, 72, 75	10	0.39	0.26-0.84	10	0
		LS	76, 78, 82					
		LS	1978, 82	31	0.50	0.18-1.31	29	6
	WF	CM	1970	1	0.16	0.16-0.16	0	0
Notigi 80/12-13 W	NP	CM	1977, 80, 82	4	1.92	1.59-2.21	100	100
		LS	1980-83	110	1.50	0.15-2.82	89	83
	YP	CM	1978, 80, 82	5	1.53	0.26-2.59	80	80
		LS	1978, 80-83	85	1.62	0.19-3.47	88	74
	WF	CM	1980	1	0.12	0.12-0.12	0	0
		LS	1980, 81, 83	92	0.21	0.06-0.71	1	0

LOCATION	SPECIES	SURVEY TYPE	YEARS	N	CONCENTRATION (ug/g)		% GREATER THAN	
					MEAN	RANGE	0.54	1.00
Numakoos 95/10 W	WF	CM	1970	1	0.20	0.20-0.20	0	0
	NP	LS	1977	18	0.16	0.07-0.41	0	0
Nutawahunan 77-78/2-3 E	WF	CM	1971	1	0.01	0.01-0.01	0	0
	NP	CM	1976, 77, 80, 82	7	0.66	0.56-0.84	100	0
		LS	1979, 80, 82	24	0.60	0.22-1.10	54	13
	YP	CM	1976, 77, 79-82	8	0.27	0.17-0.37	0	0
		LS	1979, 82	36	0.29	0.13-0.75	3	0
Opachuanau 89/14 W	NP	CM	1971, 73, 76-79	13	0.42	0.20-0.97	23	0
		LS	1978-1981	44	0.51	0.12-1.17	45	2
	YP	CM	1970, 71, 73 1975-1980	14	0.28	0.12-0.72	7	0
		LS	1976-1981	83	0.46	0.14-1.44	24	5
	WF	CM	1970, 76, 81	3	0.08	0.05-0.13	0	0
		LS	1981	5	0.14	0.10-0.19	0	0
Osik (Sawbill) 80/10 W	NP	CM	1982, 83	1	1.57	1.32-1.83	100	100
		LS	1982, 83	10	1.46	0.41-2.21	90	90
	YP	CM	1982, 83	4	0.69	0.09-1.05	75	25
		LS	1982, 83	24	0.70	0.05-1.23	71	38
Oskatuk Block 101/25 W	NP	CM	1980, 83	2	0.46	0.36-0.56	50	0
		LS	1980, 83	11	0.46	0.26-0.80	36	0
	YP	CM	1980, 83	2	0.37	0.30-0.44	0	0
		LS	1980, 83	11	0.42	0.15-0.80	27	0
Ospawagan 76/2 W	NP	CM	1978, 83	2	0.83	0.11-1.56	50	50
		LS	1983	5	1.27	0.36-1.77	80	60
	YP	CM	1978, 83	3	0.65	0.18-0.92	67	0
		LS	1982	12	0.97	0.67-1.54	100	33

LOCATION	SPECIES	SURVEY TYPE	YEARS	N	CONCENTRATION (ug/g)		% GREATER THAN	
					MEAN	RANGE	0.54	1.00
Oxford 67-68/13 E	NP	CM	1972, 77, 80, 82	8	0.21	0.15-0.31	0	0
		LS	1982	5	0.17	0.16-0.19	0	0
	YP	CM	1972, 77, 80, 82	8	0.21	0.13-0.27	0	0
		LS	1982	5	0.13	0.08-0.20	0	0
Paint 75/3-4 W	NP	CM	1975	1	0.31	0.31-0.31	0	0
		LS	1983	25	0.49	0.24-0.96	28	0
	YP	CM	1975	1	0.33	0.33-0.33	0	0
		LS	1983	11	0.40	0.24-0.54	0	0
Pakwa 67-68/10 W	WF	CM	1970	1	0.07	0.07-0.07	0	0
	NP	CM	1971, 73-78	66	0.48	0.25-1.10	27	3
		LS	1978	11	0.32	0.09-0.70	9	0
	YP	CM	1970, 71, 73 75-78, 81	24	0.27	0.18-0.42	0	0
		LS	1978, 81	12	0.30	0.15-0.51	0	0
Partridge Breast 96/3 W	NP	CM	1975, 82	3	0.66	0.37-0.92	67	0
		LS	1982	10	0.79	0.56-1.18	100	20
	YP	CM	1982	1	0.63	0.63-0.63	100	0
		LS	1982	6	0.70	0.64-0.81	100	0
Partridge Crop 76-77/1-2 E	WF	CM	1970	1	0.07	0.07-0.07	0	0
	NP	CM	1971-80	47	0.66	0.19-1.34	68	9
		LS	1978-80	15	0.60	0.20-1.31	60	7
	YP	CM	1970-80	20	0.31	0.22-0.54	0	0
		LS	1978, 80	16	0.24	0.13-0.37	0	0
Paskwachi 93/25-26 W	NP	CM	1981	1	0.38	0.38-0.38	0	0
		LS	1981, 82	10	0.44	0.23-0.73	40	0
	YP	LS	1982	5	0.44	0.29-0.60	20	0
Philips 72/6 W	NP	CM	1982	2	0.36	0.26-0.47	0	0
		LS	1982	19	0.33	0.16-0.69	11	0

LOCATION	SPECIES	SURVEY TYPE	YEARS	N	CONCENTRATION (ug/g)		% GREATER THAN	
					MEAN	RANGE	0.54	1.00
Pikwitonei 75-76/3 E	WF	CM	1970,71	3	0.11	0.03-0.25	0	0
	NP	CM	1970,71,76 78,81,82	12	0.32	0.12-0.78	17	0
		LS	1978,81,82	15	0.61	0.17-1.56	53	7
	YP	CM	1970-72 76-79,81-82	15	0.24	0.12-0.35	0	0
		LS	1978-79,80,82	22	0.27	0.11-0.57	9	0
Pine 91/14 W	NP	CM	1978,81,82	3	0.58	0.43-0.69	67	0
		LS	1977,78,81,82	35	0.40	0.06-1.26	29	6
	YP	CM	1978,82	2	0.43	0.29-0.57	50	0
		LS	1978,82	12	0.44	0.16-0.74	33	0
Pipestone 64/2 W	NP	CM	1971-73,76 1981-83	21	0.52	0.15-1.28	33	5
		LS	1981-83	53	0.59	0.12-2.59	47	8
	YP	CM	1970-73,75 76,81-83	30	0.47	0.20-0.74	27	0
		LS	1981-83	85	0.56	0.10-1.11	53	2
	WF	CM	1970		0.05	0.05-0.05	0	0
Playgreen 57/5 W	NP	CM	1970,71,78,81	16	0.35	0.21-0.54	0	0
		LS	1978,81	84	0.22	0.04-0.85	2	0
	YP	CM	1970,71 76-78,81	16	0.21	0.08-0.39	0	0
		LS	1978,81	112	0.20	0.02-0.57	1	0
	WF	CM	1970,71,78	7	0.09	0.03-0.14	0	0
		LS	1981	85	0.03	0.01-0.09	0	0
Pukatawagan 78/25 W	NP	CM	1983	1	0.56	0.56-0.56	100	0
		LS	1983	5	0.47	0.29-0.77	20	0
	YP	CM	1983	1	0.48	0.48-0.48	0	0
		LS	1983	5	0.44	0.36-0.65	20	0

LOCATION	SPECIES	SURVEY TYPE	YEARS	N	CONCENTRATION (ug/g)		% GREATER THAN	
					MEAN	RANGE	0.54	1.00
Pulse 97/10 W	NP	CM	1977	8	0.35	0.19-0.71	13	0
	YP	CM	1977	14	0.38	0.18-0.74	21	0
Rat 83/14 W	NP	CM	1978, 80	6	2.17	2.04-2.32	100	100
		LS	1978, 80, 83	47	1.82	0.30-2.49	98	96
	YP	CM	1978, 80	6	2.31	1.15-2.67	100	100
		LS	1978, 80, 83	123	1.94	0.34-4.10	96	79
	WF	CM	1978, 80	2	0.35	0.34-0.37	0	0
		LS	1978, 80, 83	52	0.31	0.13-0.70	8	0
Rat East 65-66/13-14 E	WF	CM	1971	1	0.03	0.03-0.03	0	0
	NP	CM	1971, 72, 78, 82	7	0.22	0.10-0.33	0	0
		LS	1982	10	0.23	0.12-0.48	0	0
	YP	CM	1971, 72, 78, 82	8	0.13	0.10-0.20	0	0
		LS	1982	12	0.15	0.08-0.22	0	0
Rat River 77/11 W	WF	LS	1983	17	0.58	0.16-0.98	65	0
	NP	LS	1978, 1983	45	1.99	0.66-3.86	100	98
	YP	LS	1983	12	2.35	1.58-3.29	100	100
Recluse 91/11 E	NP	LS	1982	17	0.44	0.23-0.88	29	0
	YP	LS	1982	22	0.57	0.27-1.33	45	5
Red Sucker 59-60/1-2 E	NP	CM	1971, 72, 76-80	18	0.29	0.10-0.40	0	0
	YP	CM	1971, 72, 76-80	18	0.32	0.20-0.50	0	0
Reed 65/20-21 W	WF	CM	1973	2	0.30	0.03-0.04	0	0
	NP	CM	1971-73, 79	11	0.25	0.16-0.34	0	0
		LS	1979	5	0.24	0.13-0.50	0	0
	YP	CM	1972, 73	8	0.25	0.09-0.37	0	0

LOCATION	SPECIES	SURVEY TYPE	YEARS	N	CONCENTRATION (ug/g)		% GREATER THAN	
					MEAN	RANGE	0.54	1.00
Reindeer 95/28 W	WF	CM	1970, 72	3	0.22	0.15-0.33	0	0
	NP	CM	1971, 72, 76 77, 79	9	0.36	0.04-0.69	11	0
		LS	1979, 80	10	0.63	0.42-1.02	60	10
	YP	CM	1970, 71, 72, 79	7	0.34	0.17-0.49	0	0
		LS	1979, 80	11	0.43	0.20-0.91	27	0
Richardson 73/21 E ,	NP	CM	1972	1	0.31	0.31-0.31	0	0
	YP	CM	1972	1	0.26	0.26-0.26	0	0
Robinson 62/7-8 E	NP	CM	1978, 81	5	0.31	0.28-0.35	0	0
		LS	1981	15	0.33	0.11-0.68	7	0
	YP	CM	1978, 81	3	0.39	0.22-0.63	33	0
		LS	1981	12	0.50	0.12-1.11	33	8
Rock Island 68/8 W	YP	CM	1981	1	0.45	0.45-0.45	0	0
		LS	1981	6	0.51	0.43-0.57	33	0
Rocky 59/28 W	NP	CM	1977	1	0.44	0.44-0.44	0	0
Root 58-59/27 W	NP	CM	1976	4	0.16	0.11-0.23	0	0
	YP	CM	1976	3	0.39	0.28-0.53	0	0
Rorke 64/10 E	NP	CM	1971	3	0.17	0.11-0.27	0	0
Russell 83-84/26 W	WF	CM	1970	1	0.09	0.09-0.09	0	0
	NP	CM	1971-73, 77, 82	23	0.72	0.43-1.18	78	13
		LS	1982	5	0.76	0.59-1.03	100	20
	YP	CM	1971, 73, 76-77, 82	28	0.40	0.21-0.64	11	0
		LS	1982	11	0.46	0.26-0.85	18	0

LOCATION	SPECIES	SURVEY TYPE	YEARS	N	CONCENTRATION (ug/g)		% GREATER THAN	
					MEAN	RANGE	0.54	1.00
Rusty 87/14 W	NP	CM	1982	1	0.50	0.50-0.50	0	0
		LS	1982	6	0.56	0.35-0.72	67	0
	YP	CM	1977,83	3	0.48	0.39-0.57	33	0
		LS	1983	11	0.50	0.26-1.39	27	9
Sabomin 72/2 E	NP	CM	1971,73,75	5	0.22	0.13-0.38	0	0
	YP	CM	1971,73,75,76	6	0.14	0.10-0.18	0	0
Sandy 113/10 W	NP	LS	1983	5	0.62	0.37-0.84	80	0
	YP	CM	1981	1	0.60	0.60-0.60	100	0
		LS	1981,83	16	0.66	0.34-1.18	69	6
Sawbill 99/26 W	NP	CM	1981	1	0.46	0.46-0.46	0	0
		LS	1981	10	0.64	0.24-1.28	40	30
	YP	CM	1981	4	0.78	0.65-0.97	100	0
		LS	1981	30	0.93	0.43-1.48	93	43
Seller 60/19-20 E	WF	CM	1970	1	0.06	0.06-0.06	0	0
	NP	CM	1970,71,81	3	0.32	0.18-0.42	0	0
		LS	1981	5	0.38	0.30-0.47	0	0
	YP	CM	1970,71	3	0.31	0.23-0.36	0	0
Semmens 70/22 E	NP	CM	1982	1	0.16	0.16-0.16	0	0
		LS	1982	5	0.18	0.10-0.26	0	0
Semple 69-70/12-13 E	NP	LS	1978	13	0.28	0.10-0.62	15	0
	YP	LS	1978	8	0.30	0.13-0.70	13	0
Settee 93/4 E	NP	LS	1982	28	0.63	0.26-1.48	64	11
	WF	LS	1982	6	0.12	0.08-0.15	0	0

LOCATION	SPECIES	SURVEY TYPE	YEARS	N	CONCENTRATION (ug/g)		% GREATER THAN	
					MEAN	RANGE	0.54	1.00
Setting 68-69/8-9 W	NP	CM	1970,71,73 75-79	18	0.46	0.34-0.76	11	0
		LS	1978,79	15	0.61	0.36-1.26	47	7
	YP	CM	1970,71, 73-78,80	17	0.30	0.17-0.89	12	0
		LS	1978,80	22	0.53	0.21-1.53	32	18
Silsky 75/11-12 E	WF	CM	1970	1	0.36	0.36-0.36	0	0
	NP	CM	1970,72,73 74,78	6	0.14	0.09-0.20	0	0
		LS	1978	5	0.16	0.08-0.28	0	0
	YP	CM	1970,72,73 74,78	5	0.18	0.11-0.30	0	0
		LS	1978	6	0.20	0.16-0.24	0	0
Simonhouse 64/25 W	NP	CM	1977	2	0.28	0.28-0.28	0	0
	YP	CM	1977	1	0.41	0.41-0.41	0	0
Simpson 91/19 W	NP	CM	1979	1	0.94	0.94-0.94	100	0
		LS	1979	5	0.93	0.20-1.66	80	40
	YP	CM	1979	1	0.53	0.53-0.53	0	0
		LS	1979	6	0.54	0.17-0.95	30	0
Sipiwesk 70/2 W	NP	CM	1970,74,75 1978-83	172	0.92	0.16-2.31	92	33
		LS	1972-74,76-83	1266	0.81	0.07-3.55	75	23
	YP	CM	1970,74,75 78-82	105	0.75	0.12-1.32	84	12
		LS	1972-83	862	0.79	0.07-2.28	77	24
	WF	CM	1970	3	0.17	0.13-0.21	0	0
		LS	1977,83	18	0.06	0.01-0.20	0	0

LOCATION	SPECIES	SURVEY TYPE	YEARS	N	CONCENTRATION (ug/g)		% GREATER THAN	
					MEAN	RANGE	0.54	1.00
Sisipuk 77-78/28-29 W	WF	CM	1970	1	0.05	0.05-0.05	0	0
	NP	CM	1970-75,78,81	23	0.34	0.15-1.57	13	4
		LS	1978,81	20	0.42	0.12-0.92	30	0
	YP	CM	1970-75,78,81	20	0.22	0.10-0.76	5	0
		LS	1978,81	24	0.18	0.10-0.54	0	0
Solomon 67/2-3 W	NP	CM	1973,77,79,80	5	0.44	0.19-0.82	20	0
		LS	1979,80	10	0.65	0.38-1.05	60	20
	YP	CM	1973,77,79,80	4	0.48	0.29-0.73	25	0
		LS	1979,80	11	0.56	0.27-1.01	45	9
Southern Indian 90-95/7 W	WF	CM	1970	1	0.08	0.08-0.08	0	0
	NP	CM	1971,72	4	0.30	0.11-0.44	0	0
	YP	CM	1970,72	2	0.21	0.18-0.24	0	0
Southern Indian 95/7 W	NP	CM	1971-73,76-83	136	0.75	0.24-2.04	73	16
		LS	1978-83	1512	0.70	0.05-3.05	67	13
	YP	CM	1971-73,75-83	83	0.52	0.16-1.26	43	6
		LS	1975,78-83	1117	0.55	0.06-2.81	40	7
	WF	CM	1970,72,73,80	11	0.07	0.02-0.11	0	0
		LS	1975,78-83	1069	0.16	0.01-1.04	2	0
Split 83/9 E	NP	CM	1970,78,79,82	8	0.58	0.24-1.09	50	13
		LS	1972,73,76 1978-1983	305	0.54	0.10-1.54	44	4
	YP	CM	1970,74,77-82	35	0.63	0.24-0.96	71	0
		LS	1972-74,76 1978-1983	283	0.60	0.08-2.67	52	8
	WF	CM	1970	2	0.09	0.04-0.14	0	0
		LS	1976,83	28	0.08	0.03-0.21	0	0

LOCATION	SPECIES	SURVEY TYPE	YEARS	N	CONCENTRATION (ug/g)		% GREATER THAN	
					MEAN	RANGE	0.54	1.00
Stag 86/19 W	NP	CM	1979,82,83	5	0.79	0.49-1.41	40	40
		LS	1979,82,83	25	0.74	0.32-2.14	68	20
	YP	CM	1972,77,78 1979,82,83	8	0.53	0.25-0.68	63	0
		LS	1978,79,82,83	33	0.57	0.18-0.91	55	0
Stephens 85/16 E	NP	CM	1983	45	1.41	0.37-3.66	84	76
	YP	CM	1981	1	1.76	1.76-1.76	100	100
		LS	1981,83	14	1.82	0.97-2.60	100	93
	WF	LS	1983	20	0.24	0.78-0.45	0	0
Stoney 89/15 W	NP	CM	1980	1	0.41	0.41-0.41	0	0
		LS	1980	6	0.39	0.22-0.67	17	0
	YP	CM	1980	1	0.39	0.39-0.39	0	0
		LS	1980	6	0.41	0.36-0.51	0	0
Story 88-89/23 W	NP	CM	1980	2	0.89	0.40-1.39	50	50
		LS	1980	11	0.85	0.33-1.91	55	36
	YP	CM	1980	2	0.79	0.75-0.83	100	0
		LS	1980	12	0.89	0.26-1.19	83	42
Stupart 76/22 E	WF	CM	1970	1	0.06	0.06-0.06	0	0
	NP	CM	1979-82	5	0.49	0.27-0.91	20	0
		LS	1979-82	35	0.56	0.11-1.28	40	11
	YP	CM	1971,72,79-82	10	0.40	0.32-0.52	0	0
		LS	1979-82	32	0.43	0.20-0.72	22	0
Suwannee 82/18 W	NP	CM	1970,77,79	3	0.29	0.17-0.52	0	0
		LS	1979	7	0.17	0.13-0.23	0	0
	YP	CM	1970,79	2	0.50	0.12-0.89	50	0
		LS	1979	6	0.95	0.64-1.22	100	50

LOCATION	SPECIES	SURVEY TYPE	YEARS	N	CONCENTRATION (ug/g)		% GREATER THAN	
					MEAN	RANGE	0.54	1.00
Swede 91/19 W	NP	CM	1973	2	0.28	0.27-0.30	0	0
	YP	CM	1973	1	0.41	0.41-0.41	0	0
Talbot 58/17 W	NP	CM	1971, 72, 82	28	0.38	0.13-0/76	21	0
		LS	1971, 82	61	0.27	0.11-0.82	5	0
	YP	CM	1971, 72, 78, 82	26	0.25	0.14-0.57	4	0
		LS	1971, 72, 82	15	0.32	0.10-0.80	20	0
Thorn 74/9 E	NP	CM	1981	1	0.46	0.46-0.46	0	0
		LS	1981	6	0.47	0.33-0.61	50	0
	YP	CM	1981	1	0.23	0.23-0.23	0	0
		LS	1981	6	0.23	0.14-0.32	0	0
Thorsteinson 95/1W-1E	NP	CM	1982	1	0.27	0.27-0.27	0	0
		LS	1982	6	0.22	0.10-0.49	0	0
	YP	CM	1977	1	0.45	0.45-0.45	0	0
Threepoint 77/10 W	NP	LS	1980, 81, 83	88	1.47	0.25-3.53	90	74
	YP	LS	1980, 81, 83	102	1.34	0.32-2.70	93	78
	WF	LS	1980, 81, 83	51	0.30	0.10-0.91	12	0
Tod 87/27-28 W	NP	CM	1974	1	1.47	1.47-1.47	100	100
	YP	CM	1974	1	0.46	0.46-0.46	0	0
Torrance 92-93/5 W	NP	CM	1977, 78, 83	3	0.59	0.40-0.88	33	0
		LS	1978, 83	10	0.44	0.21-0.85	30	0
	YP	CM	1977-79, 83	6	0.42	0.35-0.54	0	0
		LS	1978, 79, 83	23	0.45	0.28-0.67	30	0
Touchwood 63/17 E	WF	CM	1971	1	0.14	0.14-0.14	0	0
	NP	CM	1971, 72, 77, 78	6	0.23	0.14-0.32	0	0
	YP	CM	1971, 72, 77 1978, 81	9	0.13	0.11-0.17	0	0
		LS	1981, 82	18	0.19	0.08-0.70	6	0

LOCATION	SPECIES	SURVEY TYPE	YEARS	N	CONCENTRATION (ug/g)		% GREATER THAN	
					MEAN	RANGE	0.54	1.00
Trout 70/4 E	NP	CM	1971	1	0.47	0.47-0.47	0	0
	YP	CM	1971	1	0.70	0.70-0.70	100	0
Tullibee 72/7-8 W	YP	CM	1973, 75	2	0.36	0.30-0.42	0	0
Uhlman 88-89/6-7 W	NP	CM	1977, 79, 81	4	0.56	0.44-0.68	50	0
		LS	1979, 81	10	0.54	0.29-1.02	30	10
	YP	CM	1977, 81	2	0.53	0.48-0.58	50	0
		LS	1981	6	0.53	0.25-0.95	33	0
Utik 72/11 E	NP	CM	1971, 73, 76 1977, 81, 82	20	0.27	0.18-0.79	5	0
		LS	1981, 1982	37	0.29	0.05-1.18	5	3
	YP	CM	1970-73 75-77, 81-82	42	0.26	0.12-0.82	7	0
		LS	1981, 1982	64	0.27	0.11-0.57	3	0
	WF	CM	1970	1	0.10	0.10-0.10	0	0
Wachi Lake 98/24-25 W	NP	CM	1982, 83	5	0.72	0.44-1.20	80	20
		LS	1982, 83	25	0.67	0.20-1.43	48	24
	YP	CM	1982, 83	5	0.60	0.41-0.69	60	0
		LS	1982, 83	26	0.65	0.31-1.41	65	8
Waddie 100/4 W	NP	CM	1981	2	0.64	0.55-0.74	100	0
		LS	1981	10	0.67	0.27-1.40	50	20
	YP	CM	1981	1	0.78	0.78-0.78	100	0
		LS	1981	6	0.78	0.33-1.44	67	33
Walker 66/4 E	NP	CM	1971-72, 75-82	23	0.30	0.12-0.62	9	0
		LS	1979-1982	25	0.29	0.13-0.67	4	0
	YP	CM	1970-72 74-77, 79-82	27	0.22	0.13-0.40	0	0
		LS	1979-1982	27	0.21	0.10-0.43	0	0
	WF	CM	1970	2	0.03	0.03-0.04	0	0

LOCATION	SPECIES	SURVEY TYPE	YEARS	N	CONCENTRATION (ug/g)		% GREATER THAN	
					MEAN	RANGE	0.54	1.00
Walton 71-71/22-23 W	YP	CM	1971	1	0.24	0.24-0.24	0	0
Wapisu 79/12 W	NP	CM	1977	3	1.53	1.48-1.61	100	100
		LS	1977, 1983	91	1.60	0.32-3.42	88	76
	YP	CM	1977	3	1.33	0.80-1.81	100	67
		LS	1977, 1983	141	1.13	0.41-3.03	87	57
	WF	LS	1983	41	0.31	0.09-0.82	7	0
War 80/12 E	NP	CM	1972, 75	2	0.15	0.06-0.25	0	0
	YP	CM	1971, 75, 81	3	0.18	0.12-0.27	0	0
		LS	1981	6	0.12	0.10-0.15	0	0
Waskaiowaka 87/7-8 ER	NP	CM	1971-72	19	0.37	0.13-0.66	11	0
		LS	1978, 82	46	0.43	0.17-1.02	20	2
	YP	CM	1971, 72	8	0.21	0.07-0.34	0	0
		LS	1978, 82	73	0.46	0.16-0.96	26	0
	WF	CM	1970	1	0.08	0.08-0.08	0	0
Waspeson (Man Pt) 111-112/27 W	NP	CM	1982	1	0.46	0.46-0.46	0	0
		LS	1982	5	0.47	0.38-0.70	20	0
Wells 95/22-23 W	WF	CM	1970	1	0.11	0.11-0.11	0	0
	NP	CM	1970, 77-80, 83	10	0.53	0.45-0.72	30	0
		LS	1978-80, 83	31	0.49	0.22-0.84	32	0
	YP	CM	1977-80, 83	9	0.46	0.37-0.53	0	0
		LS	1978-80, 83	34	0.45	0.22-0.78	21	0
Whiskey Jack 108/27 W	YP	CM	1974	1	0.45	0.45-0.45	0	0
White Rabbit 69/1 E	NP	CM	1971-73, 77-80	21	0.34	0.06-0.68	14	0
		LS	1978-80	27	0.48	0.23-1.15	30	4
	YP	CM	1971, 73, 77, 78	18	0.28	0.14-0.44	0	0
		LS	1978	5	0.42	0.21-0.66	40	0

LOCATION	SPECIES	SURVEY TYPE	YEARS	N	CONCENTRATION (ug/g)		% GREATER THAN	
					MEAN	RANGE	0.54	1.00
Whitefish 75-76/5-6 E	WF	CM	1970, 71	3	0.03	0.01-0.06	0	0
	NP	CM	1981	1	0.62	0.62-0.62	100	0
		LS	1978	19	0.38	0.17-0.79	5	0
Wilkie 111/11-12 W	NP	CM	1982	1	0.46	0.46-0.46	0	0
		LS	1982	5	0.49	0.37-0.63	20	0
	YP	CM	1981, 82	2	0.57	0.50-0.64	50	0
		LS	1981, 82	10	0.63	0.40-1.32	50	10
Windy 65-66/11-12 W	NP	CM	1977	1	0.64	0.64-0.64	100	0
	YP	CM	1977	1	0.45	0.45-0.45	0	0
Winnipeg	WF	CM	1971, 72	2	0.03	0.03-0.04	0	0
		LS	1978	8	0.03	0.02-0.03	0	0
	NP	CM	1971, 72, 73	101	0.43	0.04-0.95	31	0
	YP	CM	1971-75, 78	143	0.44	0.20-1.07	20	1
		LS	1978	13	0.21	0.08-0.56	8	0
Winnipeg Ar #1 53/5 W	WF	CM	1970	1	0.15	0.15-0.15	0	0
		LS	1983	12	0.02	0.01-0.04	0	0
	NP	CM	1970, 72, 73	15	0.47	0.29-1.27	13	7
		LS	1971, 73, 74, 76 78, 80, 83	154	0.40	0.09-1.15	25	2
	YP	CM	1970, 72, 73	13	0.34	0.18-0.45	0	0
		LS	1971, 73, 74 76, 78, 80	119	0.33	0.09-0.79	9	0
Wintering 73-74/2 W	WF	CM	1970	1	0.08	0.08-0.08	0	0
	NP	CM	1970-76	15	0.35	0.24-0.46	0	0
	YP	CM	1970, 71, 73 75, 76	13	0.21	0.15-0.27	0	0

LOCATION	SPECIES	SURVEY TYPE	YEARS	N	CONCENTRATION (ug/g)		% GREATER THAN	
					MEAN	RANGE	0.54	1.00
Witchai 80/4 E	WF	CM	1970	1	0.04	0.04-0.04	0	0
	NP	CM	1972,78,80	5	0.40	0.28-0.56	20	0
		LS	1978,80	20	0.42	0.20-0.73	35	0
	YP	CM	1972,76,78,80	7	0.30	0.24-0.39	0	0
		LS	1978,80	24	0.33	0.09-0.72	13	0
Wolfpack 82/23 W	NP	CM	1978,81	3	0.33	0.20-0.50	0	0
		LS	1978,81	12	0.28	0.16-0.60	8	0
	YP	CM	1978,81	4	0.43	0.29-0.66	25	0
		LS	1978,81	25	0.41	0.13-0.80	32	0
Wood 97-98/2 W	WF	CM	1970	2	0.06	0.04-0.09	0	0
	NP	CM	1981	2	0.53	0.36-0.71	50	0
		LS	1979,81	15	0.42	0.12-1.24	33	7
	YP	CM	1970,81	4	0.45	0.18-0.65	25	0
		LS	1978,79,81	23	0.36	0.05-0.99	35	0
Woosey 67/19 W	WF	CM	1973	1	0.02	0.02-0.02	0	0
	NP	CM	1971-73, 76,77,79	9	0.39	0.35-0.43	0	0
		LS	1979	8	0.34	0.16-0.56	13	0
	YP	CM	1971-73 76,77,79	9	0.27	0.18-0.34	0	0
		LS	1979	15	0.34	0.14-0.70	13	0
Wustwatim 75-76/7-8 W	NP	CM	1982	1	1.31	1.31-1.31	100	100
		LS	1979,1982	112	0.91	0.21-5.31	89	24
	YP	CM	1970,71,73-1977,1983	10	0.49	0.25-0.84	30	0
		LS	1979-81,83	161	0.82	0.25-2.18	84	24
	WF	CM	1970	1	0.08	0.08-0.08	0	0
		LS	1981	28	0.33	0.18-0.78	7	0

LOCATION	SPECIES	SURVEY TYPE	YEARS	N	CONCENTRATION (ug/g)		% GREATER THAN	
					MEAN	RANGE	0.54	1.00
Yawning Stove 62/23 W	WF	CM	1973	2	0.04	0.03-0.06	0	0
	NP	CM	1971, 73, 74 76, 81	11	0.18	0.06-0.28	0	0
		LS	1981	5	0.22	0.14-0.41	0	0
	YP	CM	1971, 73, 74 76, 81	12	0.26	0.12-0.38	0	0
		LS	1981	5	0.31	0.10-0.47	0	0
Yellow Water River 99-100/24-25 W	NP	CM	1979	1	0.37	0.37-0.37	0	0
		LS	1979	6	0.32	0.14-0.49	0	0
	YP	CM	1979	1	0.25	0.25-0.25	0	0
		LS	1979	7	0.27	0.19-0.34	0	0

SASKATCHEWAN

LOCATION	SPECIES	SURVEY TYPE	YEARS	N	CONCENTRATION (ug/g)		% GREATER THAN	
					MEAN	RANGE	0.54	1.00
Beaver (Amisk)	WF	CM	1970,71	2	0.09	0.05-0.13	0	0
	NP	CM	1971,72	7	0.27	0.18-0.34	0	0
	YP	CM	1971,72	6	0.30	0.23-0.35	0	0
Bleasdell	YP	CM	1979,80	2	0.71	0.67-0.75	100	0
		LS	1979,80	13	0.74	0.34-1.05	85	15
Charcoal	NP	CM	1983	1	0.34	0.34-0.34	0	0
		LS	1983	5	0.35	0.28-0.42	0	0
	YP	CM	1980,83	2	0.50	0.29-0.72	50	0
		LS	1980,83	8	0.63	0.24-1.32	50	13
Chicken	NP	CM	1979	2	0.35	0.25-0.45	0	0
		LS	1979	8	0.37	0.15-0.62	13	0
	YP	CM	1971,78,79	6	0.34	0.25-0.45	0	0
		LS	1979	39	0.36	0.14-0.80	15	0
Cumberland	NP	CM	1972,74-80 82,83	23	0.78	0.24-1.67	70	22
		LS	1974-77,79-83	153	0.56	0.05-2.03	35	14
	YP	CM	1972,74-80 82,83	56	0.66	0.23-1.34	59	14
		LS	1974-83	324	0.64	0.13-2.93	50	15
Galbraith	NP	CM	1976	1	0.21	0.21-0.21	0	0
	YP	CM	1976	1	0.44	0.44-0.44	0	0
Garth	NP	CM	1982	1	1.95	1.95-1.95	100	0
		LS	1982	5	1.23	0.28-3.14	60	40
	YP	CM	1982	1	1.05	1.05-1.05	100	100
		LS	1982	5	1.20	0.72-1.49	100	80

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LOCATION	SPECIES	SURVEY TYPE	YEARS	N	CONCENTRATION (ug/g)		% GREATER THAN	
					MEAN	RANGE	0.54	1.00
Kakinagimak	NP	CM	1979	1	0.50	0.50-0.50	0	0
		LS	1979	4	0.49	0.39-0.68	25	0
	YP	CM	1979	1	0.09	0.09-0.09	0	0
		LS	1979	20	0.07	0.04-0.14	0	0
Kamatsi	NP	CM	1971,81	2	0.47	0.20-0.74	50	0
		LS	1981	26	0.67	0.32-1.37	58	15
	YP	CM	1971,81	4	0.71	0.41-0.98	75	0
		LS	1981	25	1.00	0.51-1.58	96	44
Kamuchawie	NP	CM	1981	1	0.91	0.91-0.91	100	0
		LS	1981	12	0.75	0.29-1.67	67	25
	YP	CM	1981	2	0.72	0.49-0.95	50	0
		LS	1981	14	0.97	0.57-1.36	100	43
Keep	NP	CM	1972	1	0.22	0.22-0.22	0	0
Kennedy	NP	CM	1981	1	0.69	0.69-0.69	100	0
		LS	1981	24	0.70	0.32-1.08	75	4
	YP	CM	1981	1	0.98	0.98-0.98	100	0
		LS	1981	25	1.04	0.64-1.53	100	56
Kipahigan	NP	CM	1971,72	6	0.30	0.22-0.35	0	0
	YP	CM	1971,72,75	6	0.24	0.12-0.41	0	0
Kyaska	WF	CM	1973	1	0.06	0.06-0.06	0	0
	NP	CM	1980,82	4	0.59	0.46-0.75	50	0
		LS	1980,82	14	0.58	0.16-1.00	50	0
	YP	CM	1980-82	3	0.73	0.52-1.02	67	35
		LS	1980-82	21	0.71	0.39-1.60	62	19

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LOCATION	SPECIES	SURVEY TYPE	YEARS	N	CONCENTRATION (ug/g)		% GREATER THAN	
					MEAN	RANGE	0.54	1.00
Loon	NP	CM	1971,77,78 80,81	11	0.46	0.28-0.70	27	0
		LS	1978,81	15	0.50	0.27-0.87	33	0
	YP	CM	1971,77,78,81	12	0.44	0.17-0.83	8	0
		LS	1978,81	23	0.59	0.18-1.12	43	17
Mari	WF	CM	1971	1	0.05	0.05-0.05	0	0
	NP	CM	1972	1	0.26	0.26-0.26	0	0
	YP	CM	1971,72	2	0.32	0.24-0.41	0	0
Mystic	NP	CM	1972	1	0.09	0.09-0.09	0	0
	YP	CM	1972	1	0.13	0.13-0.13	0	0
Mamew	NP	CM	1971,73,74 76,77	7	0.32	0.23-0.39	0	0
		LS	1977	7	0.34	0.11-0.55	14	0
	YP	CM	1971-74,76,77	18	0.42	0.32-0.84	11	0
		LS	1977	50	0.34	0.13-0.78	6	0
Mirond	WF	CM	1970	1	0.09	0.09-0.09	0	0
	NP	CM	1978	1	0.19	0.19-0.19	0	0
		LS	1978	5	0.18	0.13-0.23	0	0
Okipwatsikew	NP	CM	1971	1	0.44	0.44-0.44	0	0
	YP	CM	1971	1	0.36	0.36-0.36	0	0
Phelan	WF	CM	1971	1	0.03	0.03-0.03	0	0
	NP	CM	1978,79	2	0.41	0.38-0.45	0	0
		LS	1978,79	10	0.38	0.16-0.59	10	0
	YP	CM	1978	1	0.64	0.64-0.64	100	0
		LS	1978	5	0.61	0.54-0.71	80	0

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LOCATION	SPECIES	SURVEY TYPE	YEARS	N	CONCENTRATION (ug/g)		% GREATER THAN	
					MEAN	RANGE	0.54	1.00
Reindeer	WF	CM	1970,71,73	6	0.05	0.03-0.07	0	0
	NP	CM	1971-73,76 1979-83	20	0.39	0.18-0.62	10	0
		LS	1979-83	27	0.52	0.35-1.07	22	4
	YP	CM	1970-73,77 1979-83	19	0.41	0.20-0.65	16	0
		LS	1979-81,83	22	0.52	0.20-0.97	45	0
Reindeer River	NP	CM	1971	6	0.38	0.21-0.60	17	0
		LS	1982	5	0.40	0.20-0.80	40	0
	YP	CM	1971	4	0.41	0.35-0.46	0	0
		LS	1982	5	0.33	0.21-0.54	0	0
Sokatisewin	NP	CM	1972,75	7	0.56	0.32-1.18	29	14
	YP	CM	1975,76,78	4	0.45	0.43-0.46	0	0
		LS	1978	6	0.49	0.12-0.89	50	0
Tomison	NP	CM	1981	1	0.73	0.73-0.73	100	0
		LS	1981	22	0.67	0.27-1.15	68	9
	YP	CM	1981,82	2	0.86	0.75-0.98	100	0
		LS	1981,82	31	0.99	0.43-1.41	97	48
Tyrell	NP	CM	1971	1	0.37	0.37-0.37	0	0
Wapus	NP	CM	1971,81	4	0.76	0.45-1.06	75	25
		LS	1981	21	0.87	0.28-2.25	71	33
	YP	CM	1971,78,81	6	0.49	0.16-0.95	33	0
		LS	1981	35	0.70	0.26-1.66	69	14
Waskuhikan	NP	CM	1979	1	0.66	0.66-0.66	100	0
		LS	1979	5	0.59	0.39-0.76	60	0
	YP	CM	1979	1	0.73	0.73-0.73	100	0
		LS	1979	5	0.81	0.51-1.00	80	0

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LOCATION	SPECIES	SURVEY TYPE	YEARS	N	CONCENTRATION (ug/g)		% GREATER THAN	
					MEAN	RANGE	0.54	1.00
Wild Nest	NP	CM	1971,72	10	0.24	0.17-0.28	0	0
	YP	CM	1971,72	11	0.27	0.18-0.47	0	0
Wallaston	WF	CM	1970	2	0.09	0.07-0.12	0	0
	NP	CM	1971,72,82	7	0.29	0.18-0.49	0	0
		LS	1982	4	0.20	0.17-0.24	0	0
	YP	CM	1970,71,81	9	0.38	0.18-0.64	11	0
		LS	1981	5	0.72	0.45-1.00	60	0

MERCURY AVAILABILITY, MOBILIZATION AND METHYLATION IN THE CHURCHILL RIVER DIVERSION AREA

A summary report for the Canada-Manitoba Agreement on the Study
and Monitoring of Mercury in the Churchill River Diversion
Project 1.3 Mobilization and Methylation

by

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SPECIAL NOTE

This report was prepared for the Canada-Manitoba Agreement on the Study and Monitoring of Mercury in the Churchill River Diversion. Any conclusions, recommendations, or opinions expressed herein are those of the authors and are not necessarily the same as those of the Steering Committee for this Agreement.

EXECUTIVE SUMMARY

The objective of this study was to document potential sources and mobilization of mercury in flooded lake ecosystems in the area of the Churchill River diversion. The study utilized a survey approach to (1) determine actual concentrations of mercury in various source materials, (2) determine mercury concentrations in small fish populations, and (3) determine specific rates of mercury methylation and demethylation in flooded lakes.

Mercury concentrations in unflooded soil and lake sediment were highly variable, and there were no obvious relationships between these concentrations and mercury concentrations in small fish. The soil and sediment mercury concentrations in the study area were similar to those reported for other unpolluted areas of the world. Mercury concentrations in unflooded soil were quite variable among sites, and between years at a given site (implying considerable sampling variation). There was no geographic pattern to variation among sites. Because of this high inter- and intra-site variation in soil mercury concentrations, detection of the depletion of soil mercury by comparison of paired samples of unflooded and flooded soil was extremely difficult. The decrease in the concentration of total mercury in upper soil horizons was determined by comparing paired samples of flooded and unflooded soil at various sites. The average decrease was about 7% but variation between compared samples was very high and only a few comparisons were statistically significant. An experiment which examined the mercury concentration of moss placed at lake bottom sites showed no short term (13 months) changes in mercury concentrations and therefore confirmed that a significant change in soil mercury concentrations after impoundment would not be expected. The depletion of soil mercury concentrations was not found to be a useful parameter to measure in the study area because (1) it is difficult to quantify and (2) it was not related to post-impoundment increases or declines in fish mercury concentrations.

On the other hand, the rate of mercury methylation was a useful parameter to measure because the mercury concentrations in populations of small fish were positively correlated with rates of mercury methylation measured at the same locale. The concentrations of mercury in populations of small fish were consistently higher in flooded lakes as compared to unflooded lakes and appeared to be related to the amount of flooding. The rates of mercury methylation were much higher in flooded soil than rates in original lake sediment. The ratios of mercury methylation rates to methylmercury demethylation rates (M/D) were also much higher in flooded soil, even though rates of demethylation were lower in original lake sediment. Mercury concentrations in small fish were not related in a meaningful way to measured limnological parameters.

The results of this study support the hypothesis that flooding leads to higher fish mercury levels because of the methylation of naturally occurring mercury in flooded soils and terrestrial vegetation. If this hypothesis is correct, then no economically feasible mitigation or amelioration measures are possible. Only the removal of all vegetation and upper soil layers before flooding would possibly eliminate the potential for unacceptably high mercury concentrations in sport and commercial fish species. Because mitigation of reservoir fish mercury problems may not be practical, emphasis should be placed on research which will assist in predicting the severity and duration

of mercury problems in existing and proposed reservoirs. Future study should be directed towards further definition of the relationships between the concentration of mercury in fish and the ratio of the rates of mercury methylation and demethylation. Methylation rate studies may be valuable for defining a proposed reservoir's potential for future fish mercury problems. The measurement of mercury concentrations in soil and lake sediment does not appear to be a valuable avenue of research for defining the duration of fish mercury problems in existing reservoirs or for predicting fish mercury problems in proposed reservoirs.

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2 Bodaly, R.A., N.E. Strange, R.E. Hecky, R.J.P. Fudge, and C. Anema. 1987. Mercury content of soil, lake sediment, net plankton, vegetation and forage fish in the area of the Churchill River diversion, Manitoba, 1981-82. Can. Data Rep. Fish. Aquat. Sci. 610: iv + 30 p.

INTRODUCTION

Inexplicably high concentrations of mercury in commercially harvested fish species were first discovered in 1978 in lakes flooded by the Churchill River diversion. Since that time, considerable survey and monitoring work in the area has demonstrated that every lake flooded by the Churchill River diversion has undergone increases in fish mercury concentrations (where pre-impoundment data were available) or has fish mercury concentrations which are noticeably higher than in nearby unflooded lakes (Bodaly and Hecky 1979; McGregor 1980; Bodaly et al. 1984). Increases in fish mercury concentrations in Wuskwatim and Southern Indian Lakes were documented by Bodaly and Hecky (1979) and by Bodaly et al. (1984). For example, mercury concentrations in pike muscle in Southern Indian Lake sampled from the commercial catch rose from 0.26-0.32 $\mu g \cdot g^{-1}$ (wet weight) over the pre-impoundment period of 1971-1973 to 0.67-0.95 $\mu g \cdot g^{-1}$ over the post-impoundment period of 1979-1982. Increases in fish mercury concentrations have also recently been documented for Stephens Lake, the reservoir created by the Kettle dam on the Nelson River (D. Ramsey, Agassiz North Associates, Winnipeg, personal communication). Mercury concentrations in northern pike muscle in Stephens Lake after impoundment (1983-1984) averaged 1.44 $\mu g \cdot g^{-1}$, while concentrations in pike in Moose Lake, the water body flooded to create Stephens Lake, averaged only 0.20 $\mu g \cdot g^{-1}$ in 1971. Other reservoirs in the area, which were also flooded as part of the Churchill River diversion project, show higher than expected (background) mercury levels in fish. For example, mean mercury concentrations in northern pike from Rat Lake have ranged from 1.53 to 2.37 $\mu g \cdot g^{-1}$ over the post-impoundment period of 1978 to 1984.

The problem of elevated mercury concentrations in fish muscle in newly formed reservoirs is proving to be remarkably widespread throughout Canada and the world. High mercury concentrations in fish in new impoundments have recently been reported in Saskatchewan (Waite et al. 1980), Labrador (Bruce and Spencer 1979), northern Quebec (Boucher et al. 1985), in addition to northern Manitoba. Also, this phenomenon has been considered during planning for the proposed Peace River Site C development (Province of British Columbia, British Columbia Utilities Commission, 1982), the proposed Slave River development in Alberta (Province of Alberta, 1982), the proposed Jackfish extension to the Ogoki diversion project in northern Ontario (W. Carey, Ontario Hydro, Toronto, Ontario: personal communication), and other projects. High mercury concentrations in fish in new reservoirs have been reported in the U.S.A. (South Carolina - Abernathy and Cumbie 1977 and Abernathy et al. 1985; Arizona - Potter et al. 1975; Idaho - Benson et al. 1976; Illinois - Cox et al. 1979; Mississippi - Knight and Herring 1972; and Utah - Smith et al. 1974) and in Finland (Lodenius et al. 1983; Verta et al. 1986). Almost all mercury found in fish muscle is in the form of methyl mercury (Westoo 1966; Lockhart et al. 1972). Methyl mercury is a powerful and persistent neurotoxin in mammalian systems. The discovery that inorganic (mercuric) mercury could be converted to methyl mercury in lake sediments (Jensen and Jernelov 1969) provided an explanation for the presence of methyl mercury in aquatic systems. However, despite 15 years of intensive research on mercury dynamics in aquatic systems, the prediction of fish mercury levels from measureable system parameters is not yet possible, nor is it possible to predict the effects of ecosystem perturbations on fish mercury concentrations. Empirical models for the prediction of fish mercury concentrations exist (e.g. Hakanson 1980) but these models lack universal application.

Studies of mercury dynamics in freshwater systems have examined factors which affect rates of mercury methylation and factors which appear to have affected the amount of mercury found in fish. The former group of studies have isolated many variables, including sediment type, temperature, pH, redox potential, the concentration of inorganic mercury, bacterial substrate, and others, as factors which affect the rate of mercury methylation. Bacterial substrate is known to be an important factor. Rissanen et al. (1972) found that methylation rates were faster in muds which were richer in organic substances. Furutani and Rudd (1980) noted that the rate of mercury methylation in both water and sediments was increased considerably by the addition of a bacterial nutrient source. Wright and Hamilton (1982) found that rates of methylation increased dramatically with nutrient addition. Akagi et al. (1979) found that in an experimental system more methyl mercury was produced over highly organic sediments than over sediments lower in organic content. Mercury methylation rates have also been found to be positively related to temperature. Rissanen et al. (1972) found that methylation rates were much faster at 20°C than at 12°C. Similarly, Wright and Hamilton (1982) found that methylation rates were higher at 20°C than at 4°C. Limnological factors which have been related to fish mercury concentrations include degree of eutrophy, water pH, water quality (especially water hardness), and turbidity.

A number of hypotheses to explain the causes of high mercury concentrations in fish in new reservoirs have appeared in the literature. Potter et al. (1975) were the first to link reservoir creation and elevated fish mercury levels. They suggested that impoundments may be susceptible to elevated fish mercury levels because they accumulate river-borne sediment which is a source of mercury. Abernathy and Cumbie (1977) concluded that the source of mercury in fish in reservoirs was the flooded soil. They hypothesized that elevated fish mercury levels may be short-lived in reservoirs because of progressive anaerobic conditions in sediments as the reservoirs age and they also noted an apparent correlation between fish mercury levels and reservoir pH, dissolved nutrients and turbidity. In the study of an Illionis water supply reservoir, Meister et al. (1979) and Cox et al. (1979) hypothesized that the source of elevated mercury in fish in this impoundment was flooded soil and that micro-organisms methylated mercury directly from the flooded soil. These two studies noted an apparent depletion of mercury in flooded soil and related this inference to evidence of declining mercury concentrations in fish muscle. Kent and Johnson (1979) noted, in similar fashion to Potter et al. (1975), that reservoirs tend to act as traps for suspended materials with which heavy metals are associated. Bodaly and Hecky (1979) and Bodaly et al. (1984) hypothesized that fish mercury problems in northern Manitoba reservoirs were due to the bacterial methylation and mobilization of naturally occurring mercury found in flooded soil.

The Canada-Manitoba Agreement on the Study and Monitoring of Mercury in the area of the Churchill River diversion was established with the objectives of understanding the underlying causes of fish mercury problems in the Churchill River diversion area, and of assessing the significance of these problems to future water management activities. Research carried out under the Project 1.3 portion of this agreement was intended to document potential sources and potential mobilization of mercury in flooded lake ecosystems in the area of the Churchill River diversion. To do this, an attempt was made to understand the underlying causes for observed increases in fish mercury concentrations in newly flooded lakes. A number of hypotheses were tested in this study:

1. that mercury concentrations in small fish in flooded lakes were greater than in unflooded lakes;
2. that mercury concentrations in small fish in reservoirs were correlated with background mercury levels;
3. that specific rates of mercury methylation were higher over flooded substrates than over natural lake sediment;
4. that soil mercury concentrations were demonstrably depleted five years after impoundment;
5. that increased mercury concentrations in small fish were related to decreases of soil mercury concentrations;
6. that mercury concentrations in small fish were related to local limnological conditions; and
7. that mercury concentrations in small fish were related to rates of mercury methylation.

This report is a summary of two original research reports. The reader should consult these original reports to obtain detailed methods and results. These two reports are Ramlal et al. (1987) (Can. Tech. Rep. Fish. Aquat. Sci. 1490) and Bodaly et al. (1987) (Can. Data Rep. Fish. Aquat. Sci. 610). Both appear as appendices to the present report. Reference will be made to the detailed results contained in these two appendices in the following manner: tables from the Technical Report (Appendix 1) will be numbered as Table T1, T2, etc., Tables from the Data Report (Appendix 2) will be numbered as Table D1, D2, etc.

METHODS

This study utilized a survey approach to (1) determine the mercury concentrations present in small fish, flooded and unflooded soil, vegetation, lake sediment, and net plankton, (2) determine limnological conditions, and (3) determine specific rates of mercury methylation at various sites in the area of the Churchill River diversion in northern Manitoba. Field work was carried out primarily in the open water seasons of 1981 and 1982. Detailed descriptions of the methods employed, including analytical precisions and detection limits, are located in Bodaly et al. (1987) and Ramlal et al. (1986; 1987). These methods will be briefly summarized here.

In the 1981 field season, 11 flooded and 2 unflooded sites were surveyed (Fig. 1). These sites were: Granville and East Mynarski Lakes (unflooded sites) and Area 5 of Southern Indian Lake (SIL), SIL Area 4, SIL Sandhill Bay, SIL Wupaw Bay, SIL Area 6, Issett, West Mynarski, Central Mynarski, Notigi (West basin), Notigi (East basin), and Footprint Lakes (flooded sites). The following parameters were determined at each of these sites: mercury concentrations in small fish (spottail shiners and yellow perch), mercury concentrations in lake sediment, flooded soil, and unflooded soil, specific rates of mercury methylation, concentrations of dissolved ions in the water column, light penetration in the water column, mercury concentrations in various size fractions of net plankton, and other parameters.

In the 1982 field season, seven sites were surveyed (Fig. 1). Two of these sites were located on unflooded lakes (Granville and East Mynarski lakes) and five of these sites were located on flooded lakes (SIL Area 5, SIL Area 4, SIL Methyl Bay, West Mynarksi Lake and Notigi Lake (West basin)). The following parameters were determined for each of these sites: the average concentrations of mercury in populations of small fish, the average concentration of mercury in flooded and unflooded soil, and specific rates of mercury methylation. In addition, rates of mercury methylation and demethylation were determined at one site (SIL Methyl Bay) in 1983.

SUMMARY OF RESULTS

Mercury concentrations in small fish

Yellow perch and spottail shiners were collected from flooded and unflooded sites in 1981 and 1982. In both species, average mercury concentrations were usually higher in flooded lakes than in unflooded lakes. (Muscle mercury concentration in yellow perch and spottail shiners were quite closely related; the correlation coefficient between mean concentrations in the two species at six sites in 1981 was 0.65). Because similar sizes of yellow perch were collected at all sites, total mercury concentrations could be compared directly among sites (Tables D9 and D11). In both years, yellow perch from the two unflooded sites showed lower total mercury concentrations in muscle than yellow perch from any of the flooded sites (Tables D9 and D11). In 1981, mean total mercury concentrations for yellow perch from both unflooded sites (Granville Lake and East Mynarski Lake) were 0.03 $\mu g \cdot g^{-1}$ while means for five flooded sites ranged from 0.05 to 0.15 $\mu g \cdot g^{-1}$ (Table D9). In 1982, the mean total mercury concentrations from the same two unflooded sites sampled were 0.02 $\mu g \cdot g^{-1}$ while means for five flooded sites ranged from 0.06 to 0.18 $\mu g \cdot g^{-1}$ (Table D11). Because similar sizes of spottail shiners were collected at all sites in 1982, total mercury concentrations could be compared directly among sites (Table D10). Mean mercury concentrations in spottail shiner in 1982 at both unflooded sites were 0.05 $\mu g \cdot g^{-1}$ while means for eight flooded sites were 0.10 to 0.24 $\mu g \cdot g^{-1}$ (Table D10). The spottail shiners sampled in 1981 could not be compared directly because of differences in the sizes of the fish from various sites. However, comparison of statistically transformed means revealed that spottail shiners from the two unflooded sites in 1981 were two of the three lowest means (Table D8). There were some indications for both the yellow perch and the spottail shiner samples that fish muscle mercury concentrations were positively correlated with extent of flooding. Samples taken from the Notigi lakes, the most extensively flooded in the Churchill River diversion area, almost always contained the highest mercury concentrations (Tables D8-D11).

Mercury concentrations in source materials and their relationship with fish mercury concentrations

Although upper soil horizons were consistently higher in total mercury concentrations than the clay subsoil, background mercury levels in unflooded soils were quite variable. Mercury concentrations in the upper soil horizons averaged about 0.10 $\mu g \cdot g^{-1}$ and concentrations in the clay subsoil averaged about 0.03 $\mu g \cdot g^{-1}$. Variation was large among sites, between years, and within sites for the upper soil layers (moss/peat/litter layer and Ah layer). For example, the mean concentration of total mercury in upper soil horizons (unclassified as to actual composition) in 1981 samples varied from 0.052 to 0.169 $\mu g \cdot g^{-1}$, or about 3-fold. The variation within a given site for individual samples was also quite large; standard deviations ranged from 0.018 to 0.085. Variations within sites was only slightly reduced from 1981 to 1982 (the average standard deviation for moss/peat/litter horizons was 0.044 in 1981 and 0.027 in 1982 and the average standard deviation for the Ah horizon was 0.031 in 1981 and 0.028 in 1982), despite changes in the soil sampling strategy designed to reduce intra-site variability. The inter-site and intra-site variation in the mercury concentration of the clay subsoil was noticeably less; for example, standard deviations for clay subsoil mercury concentrations for the 1981 samples ranged from 0.008 to 0.031 (Table D2). Inter-year variability at a given site in average mercury concentrations was surprisingly large. For example, the total mercury concentration in the Ah horizon at the Notigi Lake, West basin site in 1981 was 0.100 $\mu g \cdot g^{-1}$ (Table D2), while at the same site in 1982 it was 0.075 $\mu g \cdot g^{-1}$ (Table D3). This inter-year variability may be due to within site variation in the characteristics and mercury content of soils.

There were no discernible patterns of regional variation in the mercury content of unflooded soil. Samples taken from Southern Indian Lake unflooded soils were generally quite similar in mercury content, on an overall basis, to samples taken from sites in the Rat River valley. For example, in 1981, the mean total mercury concentration of the moss/peat/litter horizon for five Southern Indian sites was 0.103 $\mu g \cdot g^{-1}$ while the mercury content of the same horizon for seven sites in the Rat River valley (Mynarski, Notigi, Issett and Footprint lakes; see Fig. 1) was 0.087 $\mu g \cdot g^{-1}$ (Table D2). Sites which were geographically close to each other were sometimes very similar in soil mercury content and sometimes quite different. For example, in 1981, the mercury content of the moss/peat/litter horizon at the SIL Area 5 site was 0.099 $\mu g \cdot g^{-1}$ while the mercury content of the same horizon at nearby SIL Area 4 was very similar, being 0.109 $\mu g \cdot g^{-1}$ (Table D2). On the other hand, the mercury content of the moss/peat/litter horizon for the three Mynarski lake sites in 1981 varied over two-fold from 0.055 to 0.115 $\mu g \cdot g^{-1}$ (Table D2).

Like unflooded soil mercury concentrations, there was no obvious regional pattern to lake sediment mercury concentrations. The overall mean sediment mercury concentration was 0.031 $\mu g \cdot g^{-1}$ dry weight (Table D4). Mercury concentrations at both unflooded sites were not lower than at flooded sites. Although sediment mercury concentrations at the two unflooded sites were less than the overall mean for all sites, levels at these two sites were at the mid-range for the means for the 13 sites (Table D4). The mean sediment mercury concentration at the five sites in the vicinity of Southern Indian

Lake was 0.028 $\mu g \cdot g^{-1}$ and the mean for the seven sites in the Rat River valley was 0.035 $\mu g \cdot g^{-1}$ (Table D4).

There was little indication of positive relationships between background (unflooded soil or sediment) mercury concentrations and fish mercury levels. Most correlation coefficients between unflooded soil or sediment mercury levels and mercury levels in spottail shiners or yellow perch for either the 1981 or 1982 results were not statistically significant (Tables D16 and D18). In particular, there was no indication of significant positive relationships between mercury concentrations in sediment and in fish (Table D16). Only correlations between mercury levels in unflooded Ah horizon and in yellow perch or spottail shiners were significant at the $p<0.05$ level, although correlations between fish mercury concentrations and the concentration of mercury in the upper 2 cm of soil, or the moss/peat/litter horizon (above the Ah horizon), were not significant.

Rates of mercury methylation

In 1982, specific rates of mercury methylation were determined five times over the open water season at five locations at the SIL Methyl Bay site. The five locations included three over fine-grained clay lake sediment and two over flooded terrain. Rates of methylation are referred to as "specific rates" because they measure the % of available mercuric ion methylated (Ramlal et al. 1986). Rates of mercury methylation were distinctly higher in flooded soil than in fine-grained lake sediments (Table T2.2). Rates of methylation in fine-grained sediments ranged from undetectable to 0.001% of added mercury methylated per gram sediment per hour while rates of methylation in flooded soil ranged from undetectable to 0.126% per gram per hour and averaged 0.034% per gram per hour over the period 6 July to 16 August (Table T2.2). These differences were not caused by different water temperatures in the two zones: temperatures were quite similar in flooded and unflooded sediments. Rates of mercury methylation in flooded soil varied in a seasonal pattern over the open water season. Rates were generally highest in mid-summer and lower in early and late summer (Table T2.2).

In 1983, on 19 July and 1 August, the specific rates of both mercury methylation and methyl mercury demethylation were determined in flooded soil at two locations and in fine-grained sediments at one location at the SIL Methyl Bay site. Rates of mercury methylation were, as in 1982, much higher in flooded soil than in fine-grained sediments (Table T2.4). The methylation rates in clay sediments were 0.003 and 0.001% per gram per hour on 19 July and 1 August, respectively, while rates in flooded soil ranged from 0.020 to 0.824% per gram per hour and averaged 0.104% per gram per hour. Therefore, rates in flooded soil were approximately 50x those in fine-grained sediments.

Rates of methyl mercury demethylation were also higher in flooded soil than in fine-grained sediments, but only by a factor of approximately four (Table T2.4). Rates of mercury demethylation in clay sediments averaged 0.045% per gram per hour while rates in flooded soil averaged 0.198% per gram per hour. Thus, the ratio of rates of methylation to demethylation averaged about an order of magnitude higher in flooded soil than in clay-rich sediments (Table T2-4). A survey of mercury methylation rates at five flooded and two unflooded sites was conducted on 17 August 1982. Methylation rates were found to be generally lowest at unflooded sites and higher at flooded sites. The rates of methylation at East Mynarski and Granville Lakes, the two unflooded sites, were among the three lowest values (Table T2.3). The SIL Area 4 site, an area of marginal flooding, had low rates of mercury methylation on that date. Rates of mercury methylation at West Mynarski Lake, SIL Area 5 and

Issett Lake had rates 5 to 30x those of the three lowest rates while the rate at Notigi Lake (West basin) was more than 200x greater than those of the three lowest sites (Table T2.3).

Survey results from 1981 were not extensive but generally confirmed the 1982 results. Three sites were surveyed on 25 July and 18 August 1981. Rates of mercury methylation were not detectable at Granville Lake (unflooded) and at SIL Area 6 (marginally flooded) on both dates but rates of 0.035 and 0.055% per gram per hour were measured at flooded Issett Lake on the two survey dates. Measureable rates of mercury methylation in the water column were not detected at any site. Results from these surveys of mercury methylation (and demethylation) rates indicated that substrate type was an important controlling factor. Rates of methylation (and ratios of methylation to demethylation rates) were distinctly higher in flooded soil. Rates in flooded soil were extremely variable, however, and appeared to be positively related to the amount of flooding at a given site.

Relationships between fish mercury concentrations and rates of mercury methylation

Fish muscle concentrations and specific rates of mercury methylation in flooded soil were strongly positively correlated. The correlation coefficients between log mercury methylation rates and fish mercury concentrations were 0.77 (n=7) for spottail shiners and 0.83 (n=7) for yellow perch (Fig. 2) (Data from Tables T2.3, D10 and D11). These significant relationships partially depended on the Notigi Lake (West basin) site which had a specific mercury methylation rate almost an order of magnitude higher than the next highest site. However, high correlation coefficients remain after the removal of the Notigi Lake site for both spottail shiners (r=0.64, n=6) and yellow perch (r=0.61, n=6).

Depletion of soil mercury after impoundment

Soil sampling in 1981 and 1982 was done in paired series; samples of unflooded soil were matched to samples of adjacent flooded soil. Therefore, the mercury concentrations in flooded and unflooded soil samples at a given site can be compared to detect depletions of soil mercury. Also, the moss basket experiments, referred to in the Technical report, provided experimental results on the same question.

In general, there was little indication of significant changes in the concentration of mercury levels in flooded soil five years after impoundment, but interpretation of the available data was difficult because of the large amount of intra- and inter-site variability (discussed above). The overall mean for all paired observations (flooded vs unflooded) of the apparent depletion of mercury in the upper soil horizons was 6.7% but variation was very high, with a total range of observations from 67% depletion to 271% apparent enrichment and a standard deviation of 65.3 (Tables D2, D3, D5, and D6). On the basis of calculated 95% confidence intervals, few of the possible pairwise comparisons showed significant differences in mercury concentration. Of 27 such comparisons, 23 showed differences which were not significant, 3 showed significant depletions, and one showed significant enrichment of soil mercury concentrations. The results of the moss basket experiment confirm the lack of significant changes in upper soil horizon mercury concentrations (Table T3.2). Moss was placed at lake bottom sites and subsampled over a period of about 13 months. The total mercury concentration of the moss did not change appreciably over this period.

Relationships between soil mercury depletion and fish mercury concentrations

Because it appears that the apparent magnitude of depletion or enrichment of soil mercury concentrations is due to sampling variation, rather than true depletion or enrichment, investigating the relationships between apparent depletion of soil mercury and fish mercury levels probably has little meaning. In addition, only one of 12 correlation coefficients between upper soil mercury depletion and fish mercury concentrations indicated a significant positive relationship (Tables D16 and D18).

Relationships between fish mercury concentrations and limnological conditions

The possible relationships between fish mercury levels and limnological conditions were investigated in 1981 both on the basis of individual limnological measurements (taken three times over the open water season) and on the basis of mean parameter values. The parameters which showed indications of significant correlations with fish mercury levels were water temperature, conductivity and mercury concentrations in the largest netplankton fraction (Tables D16 and D17). The latter correlations were usually positive and appeared logical because zooplankton is a common food source for both spottail shiners and yellow perch. However, these correlations may be artifacts of the sampling and analytical methods. Most analyses of the total mercury content of the largest zooplankton organisms were based on quite small samples because of the scarcity of large plankters at most sites. Total mercury analyses on small plankton samples are likely subject to large errors (D. Ramsey, Agassiz North Associates, Winnipeg, pers. comm.). In this case, the mercury content of the largest netplankton fractions were only rarely correlated with the mercury content of the smaller sized organisms and there were few significant correlations between zooplankton mercury content and other limnological parameters, except for a number of negative correlations with water temperature (see below). This may be further evidence for the unreliability of the large netplankton mercury concentration results.

The correlations between fish mercury concentrations and water temperatures were all negative (most indicated statistically significant relationships). This is surprising considering the seasonality of mercury methylation rates at the Methyl Bay sites in 1982 where rates were highest in mid-summer (see above). This seasonality implies that temperature would be a positive, not negative, controlling factor affecting the rate of supply of methyl mercury. Correlations between fish mercury concentrations and water temperatures were not due to the indirect effect of temperature on fish growth. Higher temperatures could be expected to lead to lower fish mercury concentrations because they would tend to cause increased fish growth which would dilute the total fish body burden of mercury. However, there is no indication of any relationships between fish size and muscle mercury concentrations for data sets where fish of the same age were collected at the various sites (Tables D9, D10, and D11). Correlations between small fish mercury concentrations and water conductivity were generally positive for yellow perch but not significant for spottail shiner. As noted above, yellow perch and spottail shiner mercury concentrations were positively correlated to the amount of flooding at a particular site. Water conductivity, in the area of the Churchill River diversion, is generally related to amount of flooding; however, high conductivity is not necessarily caused by flooding. Rather, the most extensively flooded areas are backwater bays and tributaries where local drainage accumulates. In the Rat River valley, this local drainage now has (and had before diversion) a higher conductivity than the Churchill River. Consequently, flooded sites in the Rat River drainage (diversion route) were higher than other areas in water conductivity. This area was also the most extensively flooded.

DISCUSSION

The background (soil and sediment) mercury concentrations in the study area are similar to other concentrations reported for unpolluted areas throughout the world, providing no indication of significant enrichment of mercury levels in soils and sediments of the study area by human activities. Sediment mercury concentrations in this study ranged from 0.009 to 0.060 $\mu g \cdot g^{-1}$ dry weight and averaged about 0.03 $\mu g \cdot g^{-1}$ mercury dry weight. Sediments in Pristine Wisconsin lakes varied from 0.05 to 0.15 $\mu g \cdot g^{-1}$ mercury (Konrad 1972). Sediment mercury levels in Yukon lakes were generally <0.01 to 0.05 $\mu g \cdot g^{-1}$ dry weight but ranged up to 0.80 $\mu g \cdot g^{-1}$ (Sherbin 1979). Sediment mercury levels in Lake Superior averaged 0.083 $\mu g \cdot g^{-1}$ (Sherbin 1979). Mercury in unpolluted lake sediments in the Nottaway, Broadback and Rupert River basins of northern Quebec averaged 0.068 $\mu g \cdot g^{-1}$ (Grisel 1979). Sediments from Lake Jocassee, North Carolina (a recently impounded reservoir) had 0.03 to 0.04 $\mu g \cdot g^{-1}$ mercury dry weight (Abernathy and Cumbie 1977). Potter et al. (1975) found that Lake Powell, Arizona (also a reservoir) averaged about 0.03 $\mu g \cdot g^{-1}$ mercury in bottom sediments. Sediment mercury levels reported in this study (Table D4) agree with those of Williamson (1980) who worked in the same geographic area. Soil mercury concentrations are also similar to those reported for other natural areas. Upper soil layers in this study were found to average about 0.1 $\mu g \cdot g^{-1}$ mercury dry weight while clay subsoils averaged about 0.03 $\mu g \cdot g^{-1}$ mercury dry weight. Jonasson (1970) reported that average mercury levels for all soils (earth's crust) were about 0.08 $\mu g \cdot g^{-1}$ and that upper soil layers were typically 5-10x concentrations in subsoil layers. Lodenius et al. (1983) found upper soil layer mercury concentrations of 0.10 $\mu g \cdot g^{-1}$ and deeper soil layer concentrations of 0.02 $\mu g \cdot g^{-1}$ in Finland while unflooded soil in

Illinois contained 0.04 to 0.15 $\mu g \cdot g^{-1}$ mercury (Cox et al. 1979). Bodaly et al. (1984) noted that the modern mercury deposition rate in Southern Indian Lake sediments was very low compared to other natural areas of the world. Although the soils and sediment have relatively low concentrations compared to materials of their type elsewhere, flooded soil will have higher mercury concentrations than pre-existing lake sediment by a factor of about three. Therefore, impoundment does increase the mean concentration of mercury in materials in contact with overlying water.

The major findings of this study point to the importance of the effect of lake impoundment, especially the flooding of terrestrial soils and the elevation of mercury methylation rates over this flooded soil, in causing elevated mercury levels in fish. The rates of mercury methylation (and methylation/demethylation ratios) and mercury concentrations were found to be much higher in flooded lake substrate than in original lake sediment. Fish mercury levels were related to rates of mercury methylation and amount of flooding but not to unflooded soil or sediment mercury concentrations or to various limnological parameters considered separately. Thus, the results of this study indicate that the concentration of source mercury which is available to be mobilized to aquatic biota in flooded lake ecosystems is relatively unimportant in determining the final concentration of methyl mercury in fish, but it may be important for enhancing the growth of methylating bacteria over non-methylating bacteria (Hecky et al. 1987). It appears that the rate of mercury methylation is related to the concentration of methyl mercury in higher biota (fish). This study confirms the hypothesis of Bodaly et al. (1984) that implicated flooding as the cause of increases in fish mercury levels in new reservoirs due to the methylation of naturally occurring mercury in flooded soils. Also, the general relationship between amount of flooding and fish mercury concentrations observed by Bodaly et al. (1984) for large fish species has been confirmed in this study for small fish species.

The lack of relationships between background mercury concentrations and fish mercury levels noted in this study is not surprising. Verta et al. (1986) also found no correlation between fish mercury concentrations and concentrations of mercury in flooded soil for 20 Finnish reservoirs. Hakanson (1984) studied seven Swedish lakes, some of which were contaminated with mercury, and found non-significant correlation coefficients between sediment mercury concentrations and pike liver mercury levels. Sediment mercury levels in Hakanson's (1984) study ranged from 0.14 to 2.35 $\mu g \cdot g^{-1}$ dry weight. Phillips (1977), in his extensive review of biological indicator organisms in aquatic systems, emphasized the lack of relationship between sediment and biota mercury levels. Hammer et al. (1982) found no relationship between either fish mercury concentrations or other biota mercury concentrations and sediment mercury content for the upper Qu'Appelle River system in Saskatchewan. Sediment mercury levels in their study ranged from 0.089 to 4.6 $\mu g \cdot g^{-1}$ dry weight. In a study of lakes in central Finland Paasivirta et al. (1983) found that although pollution had increased sediment mercury by up to 30-fold, fish mercury concentrations were highest in one natural lake. Penn (1978) studied many northern Quebec lakes and found that fish mercury levels did not reflect sediment and bedrock mercury concentrations. These findings imply that factors other than concentrations of mercury in source materials can be important in the determination of fish mercury concentrations. Based on results from the present study, the rate of mercury methylation would have to be considered as a primary modifying factor especially in flooded lake ecosystems. Other modifying factors that have been reported include oxygen levels (aerobic vs anaerobic; Olson and Cooper 1976), water hardness (Rodgers and Beamish 1983), fish oxygen consumption (Rodgers and Beamish 1981), and lake pH (Wren and MacCrimmon 1983). Most of these factors were measured in the present study but were not important in determining mercury concentrations in fish.

No significant depletion of soil mercury concentrations was detected in this study. Actual depletion may have been in the order of 5 to 10% (representing depletions of only 1 to 2% per year) but detection of differences of this magnitude is limited both by sampling intensity and analytical methods. Abernathy et al. (1985) reported similar inconclusive results in a study of flooded and unflooded soil mercury concentrations in Russell Reservoir, South Carolina. A greater understanding of the small scale variation in soil mercury concentrations would be desireable in designing more effective field sampling strategies. Also, an improvement in the analytical methods for determination of total mercury concentrations in dried soil samples would be necessary. Likewise, an experiment of the type conducted in this study where moss was placed on the lake bottom would have to be highly replicated and conducted over a much longer period of time to detect significant mercury concentration depletions. This study does not confirm work done on an Illinois impoundment by Cox et al. (1979) where significant soil mercury depletion within five years of impoundment was inferred and used to explain declines in fish mercury levels. The mercury concentration in whitefish muscle in Southern Indian Lake has been slowly declining after reaching peaks in the third year after impoundment. Mercury concentrations in predatory fish have not declined significantly over this period of time. Soil mercury concentrations have not noticeably reduced five to six years after impoundment. Determination of soil mercury depletion would therefore not be useful in predicting fish mercury level declines.

RECOMMENDATIONS

Future studies

The most profitable area of future study will likely be the further definition of the relationship between rates of mercury methylation (or methylation/demethylation ratios) and fish mercury levels. The results of the present study indicate a close relationship between rates of mercury methylation and concentrations of mercury in fish; factors other than methylation rates appeared to be much less closely related to fish mercury concentrations. Methylation rate studies may be a valuable tool for the assessment of a proposed reservoir's potential for mercury bioaccumulation problems, perhaps utilizing a standardized assay technique of substrates to be flooded. Methylation rate studies may also be valuable for defining the capacity of an existing reservoir to produce and bioaccumulate methyl mercury. The determination of mercury concentrations in unflooded and flooded soil and in lake sediment are probably not useful either for assessing the potential for mercury bioaccumulation problems in proposed reservoirs or for predicting changes in fish mercury concentrations in existing reservoirs.

Mitigation

If elevated concentrations of mercury in fish in reservoirs are due to the microbial methylation of naturally occurring mercury in flooded soil, then mitigation or amelioration of the problem will prove to be very difficult. Perhaps only the removal of all vegetation and upper soil horizons before impoundment would eliminate the problem; this measure would be very costly. Covering soil (after vegetation removal) with inorganic materials (e.g. gravel, clay) would likewise be costly. Shore erosion, which often accompanies the creation of an impoundment, would tend to reduce the effectiveness of both of the above mitigation measures. Shoreline erosion would remove inorganic material placed over soil and would continually add fresh organic substrate (Newbury and McCullough 1984).

ACKNOWLEDGEMENTS

Funding for this work was provided by the Canada-Manitoba Agreement on the Study and Monitoring of Mercury in the Churchill River Diversion and by the Department of Fisheries and Oceans. Data were collected by many people including C. Anema, S. Balchin, K. Cooper, R.J.P. Fudge, T.W.D. Johnson, D.S. MacDonnell, G.K. McCullough, B. Parker, N.E. Strange, and M.A. Treble.

We thank the following individuals for valuable reviews: G.J. Brunskill, P. Campbell, Y.K. Chau, R. Hunt, K. Lum, J. Metcalfe, L. McKerness, K.H. Mills, and J.W.M. Rudd.

We also would like to express our thanks to L. McKerness, Study Coordinator, for her assistance throughout the Study Agreement.

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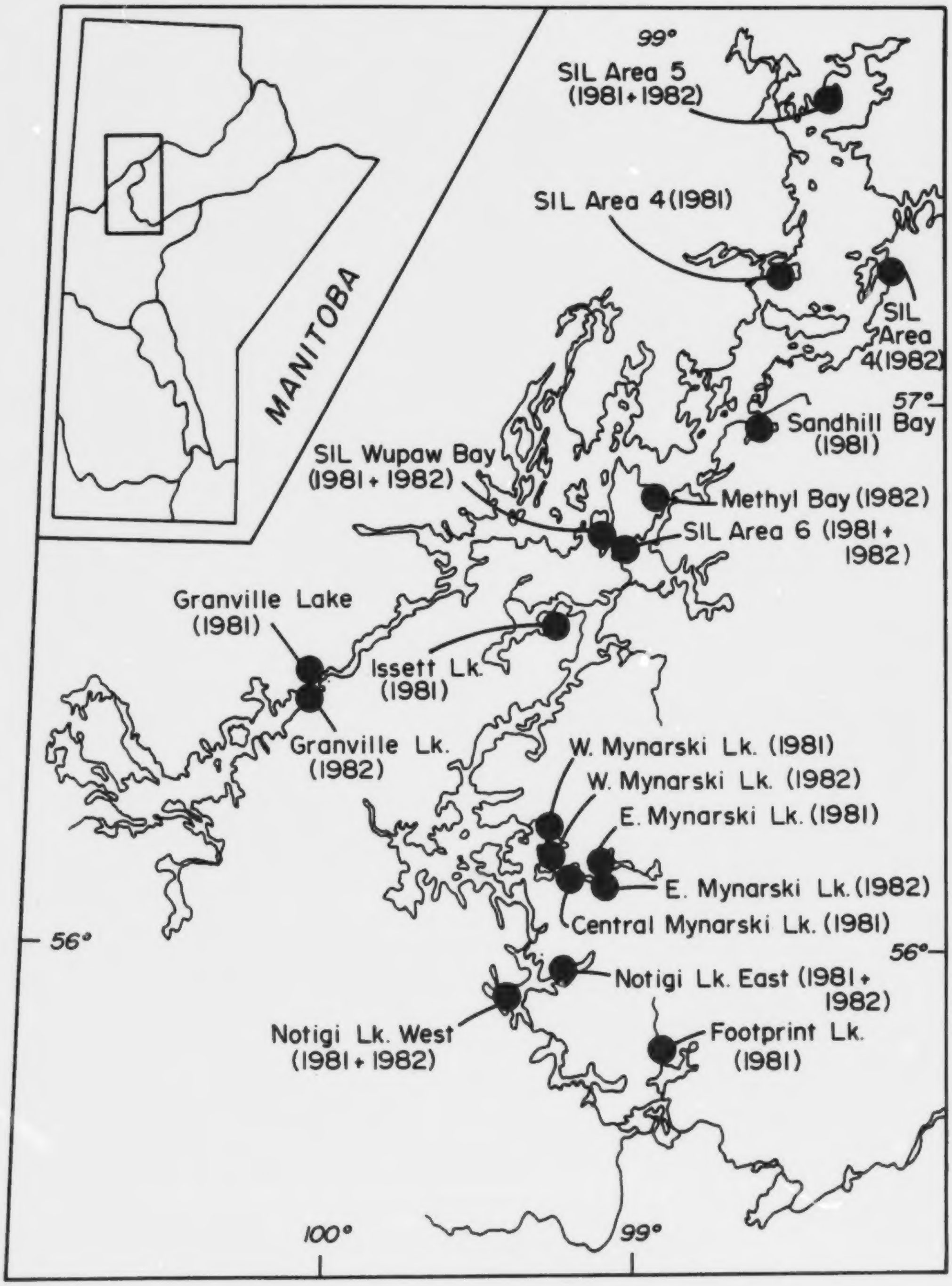
MANITOBA
99°
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57°
Sandhill Bay (1981)
SIL Wupaw Bay (1981 + 1982)
Methyl Bay (1982)
SIL Area 6 (1981 + 1982)
Granville Lake (1981)
Issett Lk. (1981)
Granville Lk. (1982)
W. Mynarski Lk. (1981)
W. Mynarski Lk. (1982)
E. Mynarski Lk. (1981)
E. Mynarski Lk. (1982)
Central Mynarski Lk. (1981)
56°
56°
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Footprint Lk. (1981)
Notigi Lk. West (1981 + 1982)
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Fig. 1 Study area showing sampling sites.

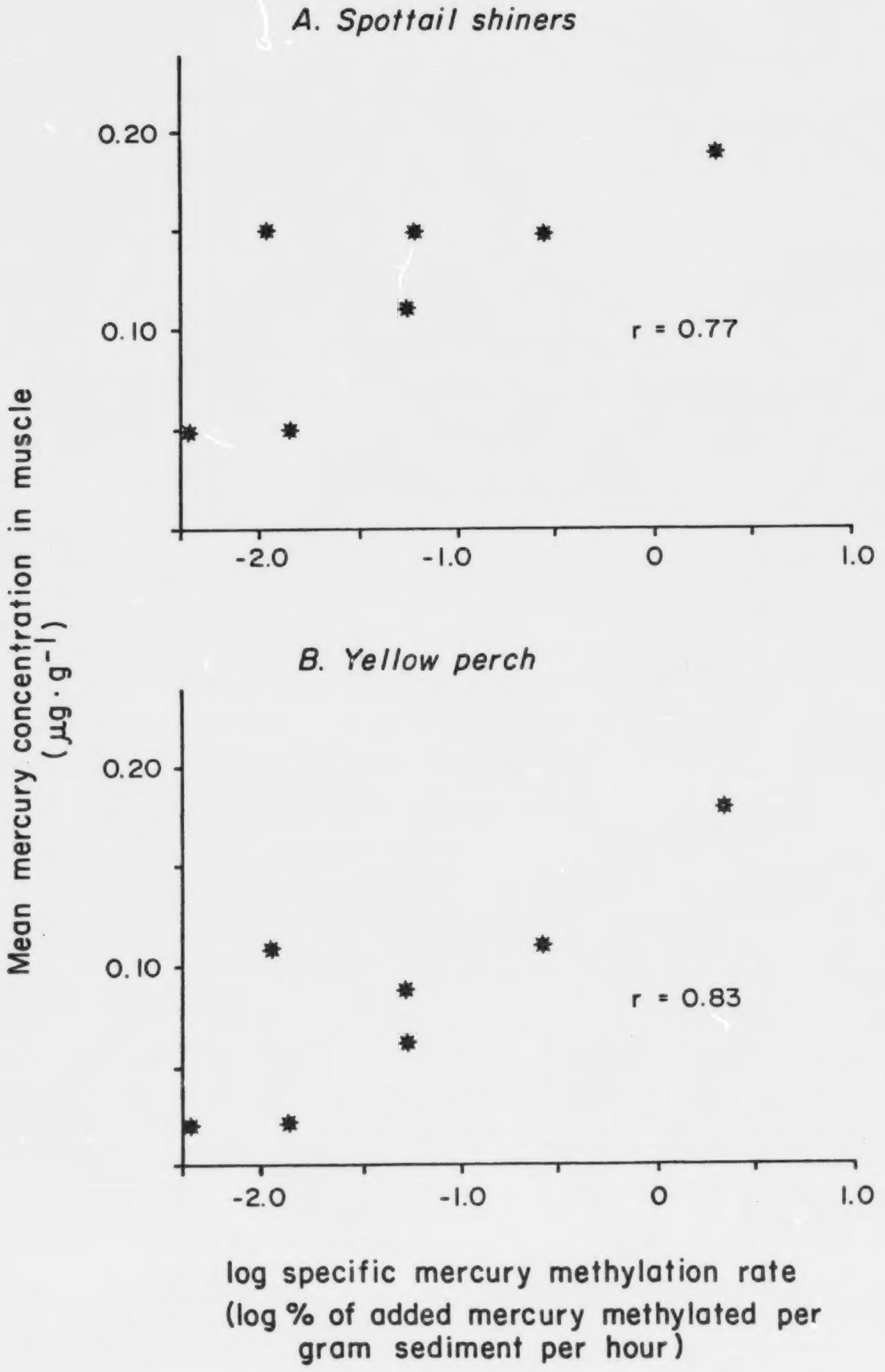

Fig. 2 Relationships between specific mercury methylation rate and spottail shiner and yellow perch mercury concentrations for seven sites in the area of the Churchill River diversion, 1982.

Canadian Data Report of
Fisheries and Aquatic Sciences 610

March 1987

MERCURY CONTENT OF SOIL, LAKE SEDIMENT, NET PLANKTON, VEGETATION, AND FORAGE FISH IN THE AREA OF THE CHURCHILL RIVER DIVERSION, MANITOBA, 1981-82

by

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This is the 3rd Data Report
from the Central and Arctic Region, Winnipeg

SPECIAL NOTE

This report was prepared for the Canada-Manitoba Agreement on the Study and Monitoring of Mercury in the Churchill River Diversion. Any conclusions, recommendations, or opinions expressed herein are those of the author and are not necessarily the same as those of the Steering Committee for this Agreement.

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Cat. no. Fs 97-13/610E ISSN 0706-6465

Correct citation for this publication is:

Bodaly, R.A., N.E. Strange, R.E. Hecky, R.J.P. Fudge, and C. Anema. 1987. Mercury content of soil, lake sediment, net plankton, vegetation, and forage fish in the area of the Churchill River diversion, Manitoba, 1981-82. Can. Data Rep. Fish. Aquat. Sci. 610: iv + 33 p.

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ABSTRACT

Bodaly, R.A., N.E. Strange, R.E. Hecky, R.J.P. Fudge, and C. Anema. 1987. Mercury content of soil, vegetation, lake sediment, net plankton and forage fish in the area of the Churchill River diversion, Manitoba, 1981-82. Can. Data Rep. Fish. Aquat. Sci. 610: iv + 33 p.

This study was undertaken to determine possible relationships between background (soil, sediment and vegetation) mercury concentrations, forage fish mercury levels, the amount of depletion of mercury in flooded soil and associated limnological parameters at various sites on and near the Churchill River diversion, northern Manitoba. In 1981, 13 sites (11 flooded, 2 unflooded) were sampled for sediment, flooded and unflooded soil, populations of small fish and limnological parameters. In 1982, seven sites (five flooded, two unflooded) were sampled for flooded and unflooded soil and populations of small fish. This report presents data for the mercury concentrations in these samples and some correlation coefficients between various quantities.

Key words: mercury; reservoirs (water); flooding; soils; moss; vegetation; sediments; spottail shiner; limnology; pH; chlorophyll; temperature (water); primary productivity; oxygen; dissolved ions.

RÉSUMÉ

Bodaly, R.A., N.E. Strange, R.E. Hecky, R.J.P. Fudge, and C. Anema. 1987. Mercury content of soil, vegetation, lake sediment, net plankton and forage fish in the area of the Churchill River diversion, Manitoba, 1981-82. Can. Data Rep. Fish. Aquat. Sci. 610: iv + 33 p.

Cette étude a été entreprise en vue de déterminer les rapports qui pourraient exister entre les concentrations de mercure de fond (sol, sédiment et végétation) le niveau de mercure dans le poisson alimentaire, la quantité de mercure que libèrent les sols inondés et les paramètres limnologiques s'y rapportant à divers endroits dans le nouveau cours du Churchill, dans le nord du Manitoba, et dans se environs immédiats. En 1981, il y a eu dans 13 lieux (11 inondés et 2 non inondés) prélèvement de sédiments, de terre dans des endroits inondés et non inondés, de sujets parmi les populations de petits poissons et contrôle des paramètres limnologiques. En 1982, il y a eu prélèvement d'échantillons de terredans des endroits inondés et non inondés ainsi que de sujets parmi les population de petits poissons dans sept lieux (5 inondés et 2 non inondés). Ce rapport présente les données sur les concentrations de mercure dans ces échantillons et des coefficients de corrélation entre diverses valeurs.

Mots-clés: mercure; réservoirs (d'eau); inondation; sols; mousse; végétation; sédiments; queue à tache noire; limnologie; pH; chlorophylle; température de l'eau; productivité primaire; oxygène; ions dissous.

INTRODUCTION

High levels of mercury in commercially important fish species were first noted in lakes flooded by the Churchill River hydroelectric project in northern Manitoba in late 1978. It was later demonstrated, for two of the lakes flooded for which both pre- and postimpoundment fish mercury data were available, that mercury levels in fish had risen coincidentally with flooding (Bodaly and Hecky 1979). The link between recently impounded reservoirs and high mercury levels in fish has recently been made in many areas (Abernathy and Cumbie 1977; Bruce and Spencer 1979; Cox et al. 1979; Potter et al. 1975; Waite et al. 1980; Lodenius et al. 1983).

An apparent relationship between the proportion of new reservoir surface area to flooded area and fish mercury levels was noted for lakes in the area of the Churchill River diversion by Bodaly et al. (1984a). This observation lead to the hypothesis that high mercury levels in fish in new reservoirs were due to the bacterial methylation of naturally occurring mercury found in flooded soils (Bodaly et al. 1984a). Cox et al. (1979) demonstrated an apparent depletion of soil mercury coincidentally with declining mercury levels in fish in an Illinois impoundment, implying that the amount of mercury available in flooded soils in reservoirs may be an important factor in determining mercury levels in fish. Abernathy and Cumbie (1977), while noting that flooded soils do provide an increased source of mercury to new reservoirs, hypothesized that declines in mercury levels in fish as reservoirs age may be related to the trophic state of the reservoir.

It is the purpose of this report to present data on mercury concentrations in forage fish, sediment, flooded and unflooded soil, vegetation, and netplankton, and to present data concerning limnological measurements which were made in conjunction with sampling for mercury content. Sampling was conducted over the course of two years at a number of sites in the area of the Churchill River diversion in northern Manitoba. Two of the sampled sites were unflooded, while the remainder were all on flooded lakes. The sampling was conducted to test the following hypotheses: first, that mercury concentrations in small fish in flooded areas was elevated relative to unflooded areas; second, that soil mercury was demonstrably depleted in Churchill River diversion reservoirs about five years after inundation; third, that mercury concentrations in populations of small fish in Churchill River diversion reservoirs were related to background (soil and sediment) mercury levels; and, fourth, that mercury concentrations in populations of small fish were related to local limnological conditions.

MATERIALS AND METHODS

STUDY SITES

The Churchill River diversion project in northern Manitoba was implemented over the period 1974-1977. About 75% of the flow of this river (or 760 $m^3 \cdot s^{-1}$) was diverted into the Nelson River basin for power production on the lower Nelson. The point of diversion was Southern Indian Lake and the diverted water followed the courses of the Rat and Burntwood River valleys before joining the Nelson River. See Bodaly et al. (1984b) and Newbury et al. (1984) for more complete descriptions of this project.

In 1981, sampling was conducted at 23 sites in the area of the Churchill River diversion (Fig. 1). Sites were chosen to encompass a broad range of flooded environments over the Churchill River diversion area and seven of the sites were adjacent to open water stations at which limnological parameters had been monitored over the period 1974-1978 (Hecky et al. 1979). Two of the sites were on lakes unaffected by the diversion: Granville Lake (coded on tables as G), on the Churchill River upstream of water control effects; and, East Mynarski Lake (EM), the eastern-most of a chain of three lakes located on a small tributary stream of the Rat River. Five sites were on Southern Indian Lake, which was raised 3 m in 1976 for the diversion of the Churchill River; Area 5 (5), Area 4 (4), Sandhill Bay (SA), Area 6 (South Bay) (6), and Wupaw Bay (WB). Five sites were part of the Notigi Reservoir in the Rat River valley, between Southern Indian Lake and the Notigi control structure. This reservoir flooded several lakes in the Rat River valley by 7-15 m over the period 1974-76: Issett Lake (I), on the upper Notigi Reservoir and raised by 7.6 m; Central Mynarski Lake (CM), the middle lake on the Mynarski chain, raised by 6.9 m; West Mynarski Lake (WM), the western-most lake on the Mynarski chain, raised by 9.0 m; the west basin of Notigi Lake (NW), and, the east basin of Notigi Lake (NE), both raised 15.2 m. One site was located on Footprint Lake (F), located on the Footprint River and flooded by 3.6 m as a result of the diverted Churchill River flows downstream of the Notigi control structure.

In 1982, sampling was conducted at seven sites (Fig. 1). Sites were chosen on the basis of the likelihood of concentrations of small fish and prevalence of mossy landscape. Six of the seven sites were on the same lake or lake basin as 1981 sites, but the precise locations usually differed from 1981; these six sites were Granville Lake, East Mynarski Lake, Area 5, Area 4, West Mynarski Lake, and Notigi Lake west basin. An additional site on Southern Indian Lake, Methyl Bay (MB), was added in 1982. Some additional spottail shiner samples were taken at the Wupaw Bay, Area 6 and Notigi Lake east basin sites.

In both years, a site was considered to consist of approximately 200 m of shoreline and adjacent areas of flooded shore (where present) and original lake bottom.

SOIL AND VEGETATION SAMPLING

Sample cores of flooded and unflooded soil were taken with modified cork borers (or similar aluminum piping) with diameters of 2.8 cm and lengths of 4 cm. The borers were fitted with plungers or pistons for core removal. The core samples were removed intact from the corer and stored frozen until analysis. Because of differences in the density of soil and moss materials cored, the actual depth of cores was variable.

In 1981, six samples were taken from each of unflooded soil, flooded soil (where present) and original lake sediment. Samples were taken at regularly spaced intervals along the length of each site.

In 1982, a set of 10 samples of unflooded soil and flooded soil (where present) was taken at each site. Each set of soil cores was taken within a small (10 m radius) area; unflooded and flooded soil samples were taken from adjacent areas. All cores were frozen within three days of recovery.

In the laboratory, frozen cores were thawed prior to separation into component soil horizon layers. A complete soil core was typically composed of three layers:

1. Moss/peat/litter - usually live green moss and/or brownish moss, sometimes lichen or broadleaf litter; structure of materials in this layer was generally distinguishable.

2. Ah horizon - brown or black colored humic rich soil horizon; vegetative structure not distinguishable.

3. Clay - brownish or gray predominately clay subsoil; sometimes contained some humus.

Not all layers were present in all cores. In particular, the clay layer was missing where the upper two layers were quite thick and the corer did not penetrate below the Ah horizon. Also, the upper moss/peat/litter layer had sometimes been eroded away by wave action from flooded soil cores.

In 1982, cores were separated into component layers according to naturally occurring divisions, as given above. In 1981, the top 2 cm of each core was removed as the first subsample; subsequent subsamples were taken according to naturally occurring divisions.

In 1981, six replicate lake sediment samples were taken at each site in a regular series adjacent to each paired sample of flooded and unflooded soil. Sediment samples were obtained using an Ekman dredge. A core was taken from the intact dredge sample to a depth of approximately 4.7 cm using a modified 50 cc disposable syringe.

Samples of representative vegetation were collected in July, 1976, as part of a study to estimate the biomass of vegetation to be flooded. Seven sites were sampled in Wupaw Bay, Southern Indian Lake (Fig. 1). The sites were chosen to represent the range of topographies, subsoils and vegetation covers that were inundated by the impoundment of Southern Indian Lake. The samples were dried after collection and subsequently analyzed for mercury content in February, 1983.

Sediment, vegetation, netplankton and soil subsamples were dried to constant weight at 100 C and then ground to a fine powder in a sample mill. Digestion of sediment and soil samples was carried out following the aqua regia procedure of Dow Chemical Co. of Canada Ltd. (Method CAS-AM-70.13). Vegetation and netplankton digestions used the potassium permanganate solution (Armstrong and Uthe 1971) to insure complete oxidation of organic matter. Total mercury content was determined by the atomic absorption spectrophotometry method of Armstrong and Uthe (1971). The analytical precision of total mercury determinations on dried material is ±0.024 at 0.6 $\mu g \cdot g^{-1}$ Hg and is ±0.010 at 0.1 $\mu g \cdot g^{-1}$ Hg while the detection limit is 0.005 $\mu g \cdot g^{-1}$.

FISH SAMPLING

Samples of small fish were collected in shallow water at each site using a seine net pulled by hand. An attempt was made to collect at least one dozen spottail shiners and one dozen yellow perch, of similar sizes, at each site. Fish were placed in plastic bags and iced immediately after capture. Samples were frozen within two days of capture and stored frozen for 2-4 months before examination. Total length was determined on frozen fish to the nearest mm. While still frozen, the body musculature was removed from both sides of each fish. Mercury concentration in the musculature was determined by the atomic absorption spectrophotometry method of Armstrong and Uthe (1971). The analytical precision of total mercury determinations on fish muscle is ±0.008 at 0.10 $\mu g \cdot g^{-1}$ Hg while the detection limit is 0.005 $\mu g \cdot g^{-1}$.

In 1981, spottail shiners collected at 12 sites ranged in size from 26 to 106 mm total length. The mercury concentration values for individual fish were adjusted for fish size using the analysis of covariance procedure of Dixon and Massey (1969). The within groups regression line was calculated using log total length (mm) as the independent variable and log total mercury concentration ($\mu g \cdot g^{-1}$ wet weight) as the dependent variable. The residuals from this line were utilized as a measure of the mercury content for individual fish. Yellow perch collected in 1981 were similar in size among sites, ranging in mean total length from 46 to 66 mm and mercury concentration was therefore not adjusted for fish size for these samples. Similarly, the mercury content of spottail shiners and yellow perch taken in 1982 was not adjusted to fish size. Spottail shiners taken from 10 of the sites in 1982 ranged in mean total length from 70 to 81 mm while yellow perch from 7 of the sites ranged in mean total length from 62 to 75 mm.

LIMNOLOGICAL SAMPLING

In 1981, a series of physical, chemical and biological parameters were determined at a station offshore of each soil sampling site (depth usually ≈5 m) three times over the open water season. Sample periods were: 1) 18-20 June, 2) 20-25 July, and 3) 17-24 August. The following parameters were determined: mean temperature of the water column (to 5 m depth) (TEMP), total dissolved nitrogen (TDN), total dissolved phosphorus (TDP), total suspended solids (TSS), suspended phosphorus (SP), suspended nitrogen (SN), suspended carbon (SC), dissolved organic carbon (DOC), percent oxygen saturation (SAT), oxygen consumption in the water column (RESP), water conductivity (COND), water column light absorbance at 455 nm after filtration and centrifugation (ABS), absorbances at 543 nm of the unfiltered sample (UNFILT), the filtered sample (FILT), and filtered/centrifuged sample (CENT), rate of primary productivity (PROD), and total mercury concentration in the zooplankton fractions >73 µm (Z73), >153 µm(Z153), >351 µm (Z351) and >760 µm (Z760). In Tables 16 and 17, values for each parameter are presented for individual sampling times (denoted as the parameter code followed by 1, 2 or 3) and for the mean of the three sample times (denoted by the parameter code preceeded by M). For water column light absorbance measurements taken at 543 nm, correlations were calculated using the change in absorbance from the unfiltered sample, UNFILT, to the filtered sample, FILT (ΔFILT) and using the change in absorbance from the unfiltered sample, UNFILT, to the filtered/centrifuged sample, CENT, (ΔCENT).

Field sampling

Temperature and conductivity profiles from surface to bottom were made using a Montedoro-Whitney model CTU-3A temperature-conductivity meter. The mean depth of appearance and disappearance of a 25 cm Secchi disk divided into alternating black and white quadrants was also recorded. Water samples for chemical analyses and primary productivity estimates were taken with a sampler (Fee 1976) which integrated between the surface and five meters depth. The water samples were taken directly into opaque polypropylene two liter bottles and stored in insulated coolers until processing at the laboratory at the South Indian Lake research station. Light extinction profiles required for the calculation of integral primary productivity were measured as described in Hecky (1984) using a Li-Cor Instruments underwater quantum sensor and meter. Two sets of twin Nitex Wisconsin nets (described by Patalas and Salki (1984)) were towed horizontally for a minimum of three minutes (longer if plankton were sparse) to collect a sufficient quantity of plankton for mercury analysis. Four different mesh sizes were used (73, 153, 351, and 760 µm) in order to sample differentially the larger components of the plankton community. The plankton samples were transferred to glass jars and stored on ice until further processing at the laboratory.

Laboratory analyses

The netplankton samples were poured onto a 10 µm mesh sieve to drain off excess water, and they were then harvested from the sieve surface into 20 mL polyethylene scintillation vials. The vials were then stored frozen until analysis for mercury content at the Freshwater Institute. Integral primary productivity (PROD) was measured using the incubator-based, numerical integration method of Fee (1973) as applied by Hecky and Guildford (1984).

Unfiltered water was used for the analysis of pH, oxygen and dissolved inorganic carbon (DIC) while filtered and centrifuged water was used for the anlaysis of NH_4^+-N. The procedures used were those of Stainton et al. (1977). Oxygen titrations were performed using an amperometric end point. The reagents for the oxygen analysis were described by Carpenter (1965). Oxygen consumption estimates (RESP) are based on two replicate initial measurements which also serve for the percent saturation estimates (SAT) and two final replicate measurements made after incubation in the dark at ambient temperatures for approximately 24 h. Oxygen consumption rates are expressed on an hourly basis.

Water samples for analysis by the Analytical Unit of the Freshwater Institute in Winnipeg were filtered through GFC glass-fiber filters and then centrifuged. Absorbance at 543 nm was measured in 10 cm cells on a Bausch and Lomb model Spectronic 70 spectrophotometer before filtration, after filtration and after centrifugation to allow estimation of suspended particulate matter. After centrifugation absorbance was measured at 455 µm for humic color (Guildford 1984). For all absorbance measurements distilled deionized water served as the reference. Suspended carbon and nitrogen were analyzed from one filter while suspended phosphorus, total suspended solids and chlorophyll were analyzed from separate filters using the methods of Stainton et al. (1977). Filtered and centrifuged water was subsampled into 20 mL plastic vials for analysis of cations (preserved with 200 mL of 1 N CHl) and into 20 mL glass vials for the analysis of DOC, (preserved with 50 µL saturated $HgCl_2$) and TDN and TDP (preserved with 100 µL of 4 N H_2SO_4). Analysis methods followed Stainton et al. (1977).

ACKNOWLEDGMENTS

We thank the many individuals who assisted in the field and laboratory: S. Balchin, K. Cooper, S.J. Guildford, T.W.D. Johnson, D.S. MacDonell, G.K. McCullough, B. Parker, A.M. Tompkins, and M. Treble. We appreciate the help of M.J. Capel, S.M. Chalanchuk, and K.E. Marshall in reviewing the manuscript.

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Table 1. Concentration of total mercury in upper 2 cm of unflooded soil (regardless of actual composition) from 13 sites in the area of the Churchill River diversion, 1981.

Site		mean total Hg conc. ($\mu g \cdot g^{-1}$ dry wt)	n	SD
5	(SIL* Area 5)	0.099	6	0.019
4	(SIL Area 4)	0.109	6	0.049
SA	(SIL Sandhill Bay)	0.110	6	0.045
WB	(SIL Wupaw Bay)	0.114	6	0.029
6	(SIL Area 6)	0.083	7	0.027
I	(Issett Lake)	0.057	6	0.036
G	(Granville Lake)	0.119	6	0.085
WM	(West Mynarski Lake)	0.115	6	0.029
CM	(Central Mynarski Lake)	0.085	6	0.018
EM	(East Mynarski Lake)	0.055	6	0.047
NW	(Notigi Lake, West basin)	0.052	6	0.046
NE	(Notigi Lake, East basin)	0.169	6	0.074
F	(Footprint Lake)	0.079	9	0.035

* SIL = Southern Indian Lake

Table 2. Concentration of total mercury in unflooded soil horizons from 13 sites in the area of the Churchill River diversion, 1981.

Horizon	Site*	Mean total Hg (μg·g^{-1} dry wt)	n	SD
moss/peat/litter	5	0.099	6	0.019
	4	0.109	6	0.049
	SA	0.110	6	0.045
	WB	0.114	4	0.027
	6	0.083	6	0.028
	I	0.057	6	0.036
	G	0.119	7	0.078
	WM	0.115	6	0.029
	CM	0.085	6	0.018
	EM	0.055	6	0.047
	NW	0.052	6	0.046
	NE	0.169	3	0.113
	F	0.079	9	0.035
Ah	5	0.115	5	0.019
	4	0.126	6	0.051
	SA	0.110	6	0.011
	WB	0.093	4	0.021
	6	0.102	4	0.009
	I	0.021	5	0.020
	G	0.080	6	0.050
	WM	0.113	3	0.032
	CM	0.108	2	0.016
	EM	0.021	4	0.023
	NW	0.100	2	0.057
	NE	0.165	4	0.037
	F	0.127	3	0.051
clay	5	0.022	2	0.012
	4	0.035	1	-
	SA	0.030	1	-
	WB	0.056	5	0.026
	6	0.027	6	0.009
	I	0.027	4	0.015
	G	0.027	3	0.015
	WM	0.050	5	0.026
	CM	0.028	6	0.008
	EM	0.033	4	0.021
	NW	0.038	6	0.031
	NE	0.035	6	0.012
	F	0.037	9	0.014

* See Table 1 for key.

Table 3. Concentration of total mercury in unflooded soil horizons from 7 sites in the area of the Churchill River diversion, 1982.

Horizon	Site		Mean total Hg ($\mu g \cdot g^{-1}$ dry wt)	n	SD
moss/peat/litter	5	(SIL Area 5)	0.115	12	0.048
	4	(SIL Area 4)	0.068	11	0.015
	MB	(SIL Methyl Bay)	0.104	10	0.021
	G	(Granville Lake)	0.156	10	0.051
	EM	(East Mynarski Lake)	0.062	5	0.014
	WM	(West Mynarski Lake)	0.207	3	0.025
	NW	(Notigi Lake West basin)	0.069	7	0.013
Ah	5		0.174	8	0.063
	4		0.075	8	0.017
	MB		0.081	10	0.012
	G		0.165	10	0.049
	EM		0.094	10	0.022
	WM		0.153	3	0.015
	NW		0.075	10	0.020
clay	5			-	
	4			-	
	MB		0.013	3	0.002
	G			-	
	EM		0.045	8	0.017
	WM		0.037	10	0.022
	NW			-	

Table 4. Concentration of total mercury in sediment samples from 13 sites in the area of the Churchill River diversion, 1981.

Site*	Mean total Hg ($\mu g \cdot g^{-1}$ dry wt)	n	SD
5	0.017	6	0.004
4	0.009	6	0.002
SA	0.014	6	0.002
WB	0.053	6	0.006
6	0.045	6	0.004
I	0.050	6	0.010
G	0.024	6	0.017
WM	0.058	6	0.013
CM	0.015	6	0.010
EM	0.020	6	0.003
NW	0.028	6	0.019
NE	0.014	6	0.003
F	0.060	6	0.004

* See Table 1 for key.

Table 5. Concentration of total mercury in flooded soil horizons from 11 sites in the area of the Churchill River diversion, 1981.

Horizon	Site*	Mean total Hg ($\mu g \cdot g^{-1}$ dry wt)	n	SD
moss/peat/litter	5	0.083	3	0.020
	4	0.067	1	-
	SA	0.088	3	0.030
	I	0.100	1	-
	NW	0.080	1	-
	F	0.094	3	0.015
Ah	5	0.060	4	0.027
	4	0.078	4	0.012
	SA	0.072	5	0.023
	WB	0.071	3	0.022
	6	0.084	6	0.028
	I	0.078	5	0.033
	WM	0.090	5	0.021
	CM	0.051	5	0.010
	NW	0.061	5	0.014
	NE	0.098	5	0.036
	F	0.073	6	0.018
clay	4	0.043	2	0.013
	SA	0.022	3	0.009
	WB	0.041	5	0.015
	6	0.035	6	0.013
	I	0.025	4	0.007
	WM	0.031	5	0.014
	CM	0.026	6	0.008
	NW	0.016	5	0.002
	NE	0.026	4	0.007
	F	0.032	9	0.021

* See Table 1 for key.

Table 6. Concentration of total mercury in flooded soil horizons from five sites in the area of the Churchill River diversion, 1982.

Horizon	Site*	Mean total Hg (μg•g^{-1} dry wt)	n	SD
moss/peat/litter	5	0.158	9	0.036
	4	0.080	6	0.012
	MB	0.117	8	0.019
	WM	0.068	6	0.016
	NW	0.047	1	-
Ah	5	0.148	6	0.012
	4	0.064	5	0.010
	MB	0.130	10	0.024
	WM	0.089	9	0.019
	NW	0.069	10	0.022
clay	MB	0.082	2	0.054
	WM	0.034	8	0.012
	NW	0.031	9	0.006

* See Table 3 for key.

Table 7. Mercury, carbon, phosphorus and nitrogen content of unflooded vegetation from Wupaw Bay, Southern Indian Lake, 1976. Replicate analyses are given for some nitrogen determinations. See Fig. 1 for site locations.

Species	Part	Sample no.	Total Hg ($\mu g \cdot g^{-1}$)	Carbon ($\mu g \cdot mg^{-1}$)	Phosphorus ($\mu g \cdot mg^{-1}$)	Nitrogen ($\mu g \cdot mg^{-1}$)	Site	Identity	Height (m)	Diameter at 1.4 m (cm)	Description or sample location on tree
Picea mariana	wood	87	<0.005	514	0.09	0.51, 0.55	WB4	TK	2.2	2.2	•lower bole
		98	0.008	492	0.07	0.23, 0.41	WB1	T2	16.0	21.8	•bole at 8 m
	bark	88	0.019	542	0.21	1.54	WB4	TK	2.2	2.2	•lower bole
		100	0.023	521	0.32	2.37	WB1	T1	8.5	7.6	•bole at 0.1 m
		99	0.029	506	0.35	2.60, 2.34	WB2	T8	6.5	8.1	•branches at 1 to 1.2 m
	twigs	19	0.070	491	0.43	4.14	WB1	T4	2.8	2.9	•0 to 1 m
		20	0.039	502	0.59	3.87	WB1	T2	16.0	21.8	•7.8 to 8 m
		9	0.039	409	0.45	-	WB4	T16	4.7	4.8	-
	foliage	21	0.014	501	0.50	5.26	WB4	T15	2.0	1.9	•1 to 1.2 m
		22	0.012	453	0.68	4.20, 4.50	WB2	T8	6.5	8.1	•1 to 1.2 m
		24	0.065	457	0.66	5.11	WB1	T4	2.8	2.9	•0 to 1 m
		25	0.010	304	0.68	5.86	WB1	T1	8.5	7.6	•7 to 7.2 m
		23	0.010	462	0.69	5.63	WB1	T2	16.0	21.8	•7.8 to 8 m
		8	0.011	334	0.52	-	WB4	T16	4.7	4.8	-
Populus tremuloides	wood	94	<0.005	533	0.03	0.48	WB6	T30	10.6	12.4	•bole at 0.2 to 0.3 m
	bark	92	0.018	505	0.30	2.41, 2.67	WB5	T29	4.3	3.4	•bole at 0 to 0.2 m
		93	0.011	328	0.78	3.13	WB6	T30	10.6	12.4	•bole at 5 to 5.1 m
	twigs	30	0.014	491	0.48	5.66, 6.36	WB5	T29	4.3	3.4	•3 to 3.2 m
		32	<0.005	503	0.66	4.78	WB6	T30	10.6	12.4	•5 to 5.2 m

Table 7. Cont'd.

Species	Part	Sample no.	Total Hg ($\mu g \cdot g^{-1}$)	Carbon ($\mu g \cdot mg^{-1}$)	Phosphorus ($\mu g \cdot mg^{-1}$)	Nitrogen ($\mu g \cdot mg^{-1}$)	Site	Identity	Height (m)	Diameter at 1.4 m (cm)	Description or sample location on tree
Populus tremuloides	foliage	31	0.011	485	1.43	18.36	WB5	T29	4.3	3.4	•3 to 3.2 m
		33	0.013	486	1.73	17.58	WB6	T30	10.6	12.4	•5 to 5.2 m
		11	0.013	223	1.54	-	WB5	T28	7.2	7.8	-
Populus balsamifera	wood	84	0.005	459	0.21	2.05	WB11	T52	6.9	6.4	•bole at 2.7 m
	bark	85	0.018	499	0.84	5.14, 6.58	WB11	T52	6.9	6.4	•bole at 2.7 m
		86	<0.005	514	1.16	6.71	WB11	T51	10.6	13.3	•bole at 9.1 m
	twigs	60	0.014	463	1.08	5.76, 8.76	WB11	T52	6.9	6.4	•at 5.5 m
		62	0.014	492	0.79	4.64, 6.68	WB11	T52	6.9	6.4	•at 6.4 to 6.6 m
	foliage	61	0.020	443	2.38	3.41, 4.01	WB11	T52	6.9	6.4	•at 5.5 m
		63	0.023	442	2.40	5.71	WB11	T51	10.6	13.3	•6.4 to 6.6 m
		64	0.027	436	3.47	8.64, 9.92	WB11	T51	10.6	13.3	•8.2 to 8.4 m
Betula papyrifera	wood	91	0.015	333	0.13	0.77, 0.93	WB2	T1	8.5	9.4	•bole at 0.3 to 0.4 m
	bark	89	0.008	545	0.26	3.61	WB2	T2	5.5	5.4	•bole at 2 to 2.1 m
		90	0.005	560	0.31	2.80, 2.98	WB3	T12	6.4	6.2	•bole at 7.6 cm diameter
	twigs	50	0.034	490	0.37	1.85, 2.47	WB2	T2	5.5	5.4	•3 to 3.2 m
		52	0.024	484	0.56	1.44, 2.60	WB3	T12	-	-	•4.1 to 4.3 m
	foliage	51	0.012	463	1.49	6.67, 8.13	WB2	T2	5.5	5.4	•3 to 3.2 m
		53	0.016	479	1.09	4.89	WB3	T12	6.4	6.2	•4.1 to 4.3 m

Table 7. Cont'd.

Species	Part	Sample no.	Total Hg ($\mu g \cdot g^{-1}$)	Carbon ($\mu g \cdot mg^{-1}$)	Phosphorus ($\mu g \cdot mg^{-1}$)	Nitrogen ($\mu g \cdot mg^{-1}$)	Site	Identity	Height (m)	Diameter at 1.4 m (cm)	Description or sample location on tree
Pinus banksiana	wood	95	<0.005	489	0.02	0.38	WB3	T14	6.2	7.6	•bole at 2.6 to 2.7 m
	bark	97	0.018	552	0.19	2.04	WB3	T14	6.2	7.6	•bole at 1.5 to 2.5 m
		96	0.013	512	0.21	2.09	WB3	T14	6.2	7.6	•bole at 2.6 to 2.7 m
	twigs	27	0.012	520	0.48	3.99	WB3	T13	6.5	7.8	•3 to 3.2 m
		29	0.007	509	0.40	3.47	WB3	T14	6.2	7.6	•5 to 5.2 m
	foliage	26	<0.005	470	0.62	5.86	WB3	T13	6.5	7.8	•3 to 3.2 m
		28	<0.005	507	0.59	6.58	WB3	T14	6.2	7.6	•5 to 5.2 m
Salix sp.[a]	wood	81	<0.005	510	0.45	1.42, 1.84	WB4	W24.4	2.5	2.2	•large branch
			-	481	0.28	-	-	R	-	-	-
	bark	83	0.016	404	0.43	4.11, 4.83	WB2	T7	1.9	1.8	•bole near base
		82	0.012	497	0.75	9.8, 11.2	WB4	W24.4	2.5	2.2	•large branch
Salix sp.	twigs	57	<0.005	465	1.17	4.34, 5.20	WB4	W18	1.8	1.1	•0.5 m to top
Salix cordata		59	0.017	427	0.77	2.91, 5.45	WB2	T5	4.7	3.0	•2.5 m to top
Salix sp.	foliage	56	0.012	445	2.39	14.7, 24.3	WB4	W18	1.8	1.1	•0.5 m to top
		10	0.017	278	1.53	-	WB4	W19	3.1	2.2	-
		54	0.010	428	1.56	15.6, 26.2	WB4	W27.3	1.9	1.8	-
		55	0.012	385	2.85	15.6, 18.0	WB2	02.2	0.7	-	-
Salix cordata		58	0.012	430	1.86	6.78, 7.60	WB2	T5	4.7	3.0	•2.5 m to top

Table 7. Cont'd.

Species	Part	Sample no.	Total Hg ($\mu g \cdot g^{-1}$)	Carbon ($\mu g \cdot mg^{-1}$)	Phosphorus ($\mu g \cdot mg^{-1}$)	Nitrogen ($\mu g \cdot mg^{-1}$)	Site	Identity	Height (m)	Diameter at 1.4 m (cm)	Description or sample location on tree
Alnus sp.	wood	78	0.006	483	0.24	1.99, 2.85	WB2	05.3	2.7	-	•large branch
	bark	79	0.010	657	0.56	10.1	WB2	05.3	2.7	-	•large branch
		80	0.011	484	0.30	7.19, 8.29	WB3	02	-	-	•small branch
	twigs	75	0.013	536	1.50	6.02, 11.8	WB2	01.2	0.4	-	-
		76	0.014	518	1.00	12.5, 13.1	WB2	04.2	1.4	-	-
		65	0.016	505	0.74	9.2, 11.4	WB2	05.3	2.7	-	-
		67	0.012	524	0.37	7.68, 9.04	WB3	02	-	-	-
	foliage	3	0.010	316	1.68	-	WB2	05.3	-	-	-
		74	0.026	467	2.88	32.1	WB2	01.2	0.4	-	-
		77	0.018	443	1.57	28.5	WB2	04.2	1.4	-	-
		66	0.013	437	1.64	21.0, 25.6	WB2	05.2	2.2	-	-
		68	0.017	466	0.77	13.6, 21.0	WB3	02	-	-	-
Ledum groenlandicum	twigs	7	0.013	408	0.21	-	WB3	01	-	-	-
		70	0.023	538	0.56	5.12, 5.52	WB3	02	-	-	-
		71	0.015	469	0.50	4.78	WB4	07	-	-	-
	foliage	6	0.015	376	1.16	-	WB3	01	-	-	-
		69	0.015	531	1.29	13.9	WB3	02	-	-	-
		72	0.020	469	1.12	13.7	WB4	07	-	-	-
		73	0.022	265	1.02	12.7	WB4	05	-	-	-
Sphagnum spp.		12	0.034	328	0.53	-	WB7	02	-	-	-
		46	0.13	448	0.65	8.51	-	-	-	-	-
		47	0.054	428	0.48	4.73	WB1	015	-	-	-
		48	0.022	398	0.27	4.78	WB4	03	-	-	-
		49	0.047	411	0.34	2.05	WB4	03	-	-	-

Table 7. Cont'd.

Species	Part	Sample no.	Total Hg (μg·g^{-1})	Carbon (μg·mg^{-1})	Phosphorus (μg·mg^{-1})	Nitrogen (μg·mg^{-1})	Site	Identity	Height (m)	Diameter at 1.4 m (cm)	Description or sample location on tree
Feather mosses		42	0.093	411	0.91	6.32	WB1	Q8	-	-	-
		43	0.053	440	1.16	7.91	WB6	Q10	-	-	•green
		44	0.073	412	0.66	4.99, 5.33	WB4	-	-	-	-
		45	0.010	396	1.04	5.71	WB1	-	-	-	-
Litter and organic soil horizon		1	0.064	257	0.67	-	WB1	-	-	-	-
		4	0.083	195	1.00	-	WB2	Q1	-	-	-
			-	384	0.86	14.70	WB2	Q4	-	-	-
		35	0.046	446	0.89	16.01	WB2	Q5	-	-	-
		36	0.039	483	0.45	8.60	WB3	Q2	-	-	-
		37	0.058	481	0.66	8.87	WB6	-	-	-	-
		38	0.059	474	0.92	8.36	WB1	-	-	-	-
		39	0.093	449	0.67	9.96, 10.84	WB1	Q8	-	-	•dominately *Picea* twigs and feather mosses •with considerable feather mosses
			-	466	0.57	12.60	WB4	Q1	-	-	•no *Sphagnum*
		41	0.094	473	0.78	9.34	WB7	Q1	-	-	-

[a] *Salix* sp.: diameter measured at 0.5 m above base (except *Salix cordata*).

Table 8. Concentration of total mercury in spottail shiner muscle from 12 sites in the area of the Churchill River diversion, 1981.

Site*	n	Mean total mercury conc ($\mu g \cdot g^{-1}$) (range)	Mean total length of fish sample (mm) (range)	Residual from "within groups" regression of log [Hg] vs length (SD)
5	12	0.38 (0.27-0.46)	94.1 (88-106)	+0.333 (0.064)
4	9	0.19 (0.16-0.25)	47.4 (40-56)	+0.186 (0.085)
SA	8	0.12 (0.11-0.14)	65.9 (60-71)	-0.076 (0.045)
WB	12	0.06 (0.05-0.07)	47.2 (40-62)	-0.315 (0.065)
6	12	0.13 (0.07-0.19)	51.5 (44-59)	-0.004 (0.128)
G	12	0.06 (0.05-0.08)	42.2 (40-45)	-0.282 (0.057)
WM	10	0.14 (0.09-0.23)	81.6 (73-87)	-0.083 (0.110)
CM	12	0.13 (0.11-0.18)	79.0 (74-85)	-0.084 (0.066)
EM	9	0.08 (0.06-0.11)	54.3 (44-71)	-0.218 (0.071)
NW	8	0.18 (0.16-0.20)	30.6 (29-32)	+0.259 (0.034)
NE	8	0.19 (0.17-0.24)	29.0 (26-31)	+0.294 (0.061)
F	18	0.17 (0.08-0.26)	60.9 (37-80)	+0.085 (0.102)

* See Table 1 for key.

Table 9. Concentration of total mercury in yellow perch muscle from seven sites in the area of the Churchill River diversion, 1981.

Site*	n	Mean total mercury conc ($\mu g \cdot g^{-1}$) (range)		Mean total length of fish sample (mm) (range)	
5	8	0.08	(0.03-0.18)	49.0	(38-65)
I	12	0.05	(0.03-0.07)	60.2	(55-65)
G	8	0.03	(0.02-0.04)	66.0	(58-90)
WM	12	0.07	(0.06-0.11)	53.5	(48-59)
CM	12	0.08	(0.06-0.10)	55.8	(51-60)
EM	12	0.03	(0.03-0.04)	46.0	(42-55)
F	24	0.15	(0.10-0.18)	55.2	(47-60)

* See Table 1 for key.

Table 10. Concentration of total mercury in spottail shiner muscle from 10 sites in the area of the Churchill River diversion, 1982.

Site*	n	Mean total mercury conc ($\mu g \cdot g^{-1}$)	(range)	(SD)	Mean total length of fish sample (mm)	(range)
5	12	0.15	(0.12-0.20)	(0.02)	70.2	(64-80)
4	12	0.15	(0.09-0.21)	(0.04)	72.3	(60-79)
MB	12	0.11	(0.05-0.16)	(0.03)	71.2	(66-79)
WB	5	0.10	(0.09-0.11)	(0.01)	81.2	(67-93)
6	12	0.10	(0.07-0.15)	(0.02)	72.7	(65-84)
G	12	0.05	(0.03-0.07)	(0.01)	74.2	(68-79)
WM	12	0.15	(0.11-0.20)	(0.03)	74.6	(66-86)
EM	12	0.05	(0.03-0.06)	(0.01)	70.8	(63-79)
NW	12	0.19	(0.16-0.24)	(0.02)	75.1	(71-80)
NE	12	0.24	(0.12-0.32)	(0.06)	76.2	(72-81)

* See Table 3 for key.

Table 11. Concentration of total mercury in yellow perch muscle from seven sites in the area of the Churchill River diversion, 1982.

Site*	n	Mean total mercury conc (μg·g^{-1}) (range) (SD)			Mean total length of fish sample (mm) (range)	
5	12	0.09	(0.06-0.13)	(0.02)	62.3	(58-70)
4	7	0.11	(0.08-0.15)	(0.03)	64.3	(62-70)
MB	12	0.06	(0.03-0.11)	(0.02)	62.5	(58.72)
G	12	0.02	(0.01-0.03)	(0.01)	71.1	(65-77)
WM	12	0.11	(0.06-0.27)	(0.06)	65.5	(57-72)
EM	12	0.02	(0.01-0.04)	(0.01)	74.8	(64-81)
NW	12	0.18	(0.14-0.25)	(0.04)	65.7	(64-70)

* See Table 3 for key.

Table 12. Major dissolved constituents at 1981 sampling sites; Na, K, Ca, Mg, Cl, SO_4, dissolved inorganic carbon (DIC), pH and specific conductance (COND) at 13 sites in the area of the Churchill River diversion, 1981. The analytical number (Anal #) is an internal reference number.

Location	Date	Anal#	Na	K	Ca	Mg	Cl	SO_4	DIC	pH	COND
						µm/L					µS/cm
G	19-Jun	35	84	21.3	179	92	35	242	576	NA	112
	25-Jul	111	96	24.4	171	98	20	36	686	7.83	66
	18-Aug	207	107	24.3	185	108	17	80	585	8.01	68
MEAN			96	23.4	179	100	24	119	616	7.92	82
6	19-Jun	33	101	22.4	216	116	25	39	747	NA	83
	25-Jul	113	105	26.5	251	121	23	36	952	8.08	85
	18-Aug	208	101	23.8	248	121	20	39	772	8.08	83
MEAN			102	24.2	238	119	23	38	824	8.08	84
4	19-Jun	31	89	20.8	377	152	NA	NA	1130	NA	122
	25-Jul	109	92	22.8	364	152	20	36	1310	8.12	116
	18-Aug	205	79	19.8	375	160	11	61	987	8.17	116
MEAN			86	21.1	372	155	16	49	1142	8.15	118
5	19-Jun	30	61	16.7	327	141	23	32	992	NA	100
	25-Jul	108	70	17.4	311	135	17	36	1050	7.86	94
	18-Aug	204	69	18.7	335	154	17	35	780	7.97	102
MEAN			67	17.6	325	143	19	35	941	7.92	99
SA	19-Jun	32	44	17.2	243	98	28	36	679	NA	76
	25-Jul	110	64	20.3	282	119	14	37	944	7.80	88
	18-Aug	206	68	19.2	314	133	11	72	734	8.18	95
MEAN			59	18.9	280	117	18	49	786	7.99	86

Table 12. Cont'd.

Location	Date	Anal#	Na	K	Ca	Mg	Cl	SO₄	DIC	pH	COND
						µm/L					µS/cm
NW	18-Jun	24	103	23.4	253	129	28	37	860	NA	87
	24-Jul	107	106	25.2	235	117	25	35	807	NA	85
	17-Aug	195	107	23.2	264	131	20	40	812	8.08	84
MEAN			105	24.0	251	126	24	37	826	NA	85
NE	18-Jun	25	101	22.9	280	131	11	229	962	NA	112
	24-Jul	106	108	26.5	251	123	23	37	845	NA	85
	17-Aug	196	109	23.2	245	125	20	39	783	8.13	84
MEAN			106	24.2	259	126	18	102	863	NA	94
F	18-Jun	26	88	25.5	536	200	11	233	1560	NA	160
	24-Jul	105	103	28.1	530	204	20	45	1420	NA	156
	17-Aug	194	104	31.1	541	212	17	46	1460	8.40	156
MEAN			98	28.3	536	205	16	108	1480	NA	157
EM	18-Jun	27	73	22.4	446	168	11	235	1300	NA	137
	23-Jul	101	83	24.0	438	173	11	87	1520	6.95	NA
	17-Aug	197	84	21.5	396	162	11	40	1140	8.33	122
MEAN			80	22.6	427	168	11	121	1320	7.64	130
CM	18-Jun	28	79	26.5	438	177	23	39	1400	NA	134
	23-Jul	102	87	28.5	433	179	8	96	1690	6.97	NA
	17-Aug	198	90	27.2	443	193	11	41	1250	8.31	133
MEAN			85	27.4	438	183	14	58	1447	7.64	134

Table 12. Cont'd.

Location	Date	Anal#	Na	K	Ca	Mg	Cl	SO$_4$	DIC	pH	COND
						µm/L					µS/cm
WM	18-Jun	29	83	28.1	433	177	25	39	1460	NA	139
	23-Jul	103	88	30.2	462	191	8	74	1640	6.95	NA
	17-Aug	199	93	27.7	425	189	17	39	1200	8.30	128
MEAN			88	28.7	440	186	17	50	1433	7.63	134
I	19-Jun	34	90	42.7	396	189	42	247	1300	NA	136
	25-Jul	112	92	43.5	375	187	14	37	1410	7.78	120
	18-Aug	209	97	46.4	385	191	20	79	1280	8.12	122
MEAN			93	44.2	385	189	25	121	1330	7.95	126
WB	20-Jun	43	90	21.3	219	112	20	49	707	NA	81
	20-Jul	92	99	22.8	237	117	20	37	831	NA	90
	24-Aug	237	105	24.9	266	129	20	39	870	7.93	89
MEAN			98	23.0	241	119	20	42	803	NA	87

Table 13. Dissolved and suspended constituents at 1981 sampling sites; Fe, Mn, total dissolved nitrogen (TDN), total dissolved phosphorous (TDP), NH_4, dissolved organic carbon (DOC), suspended nitrogen (SN), suspended phosporous (SP) and chlorophyll-a (Chl-a) at 13 sites in the area of the Churchill River diversion, 1981. The analytical number (Anal #) is an internal reference number.

Location	Date	Anal#	Fe	Mn	TDN	TDP	NH_4	DOC	SC	SN	SP	Chl-a
			µm/L									µg/L
G	19-Jun	35	1.13	<0.36	21	0.19	0.10	1980	29.7	3.4	0.26	2.3
	25-Jul	111	1.10	<0.36	NA	NA	0.33	560	37.8	1.5	0.32	4.3
	18-Aug	207	<0.36	<0.18	NA	NA	0.23	970	48.5	6.7	0.37	3.7
MEAN			<0.86	<0.30	NA	NA	0.22	1170	38.7	3.9	0.32	3.4
6	19-Jun	33	0.75	<0.36	20	0.13	0.11	680	27.0	3.4	0.27	1.5
	25-Jul	113	2.20	<0.36	NA	NA	0.53	530	28.0	2.6	0.39	4.7
	18-Aug	208	<0.36	2.68	NA	NA	0.20	550	25.3	3.8	0.32	1.8
MEAN			<1.10	<1.14	NA	NA	0.28	587	26.8	3.3	0.33	2.7
4	19-Jun	31	0.75	<0.36	23	0.39	0.25	490	21.7	2.4	0.34	2.1
	25-Jul	109	1.65	<0.36	NA	NA	0.32	600	28.7	4.1	0.28	3.0
	18-Aug	205	<0.36	<0.18	NA	NA	0.20	1150	26.0	3.9	0.30	2.0
MEAN			<0.92	<0.30	NA	NA	0.26	747	25.4	3.5	0.31	2.4
5	19-Jun	30	0.38	0.36	23	0.13	0.33	640	58.2	5.8	0.28	2.8
	25-Jul	108	1.10	<0.36	NA	NA	0.36	670	35.7	6.0	0.36	3.1
	18-Aug	204	<0.36	<0.18	NA	NA	0.26	660	41.2	5.7	0.30	3.3
MEAN			<0.61	<0.30	NA	NA	0.32	657	45.0	5.8	0.31	3.1
SA	19-Jun	32	1.50	<0.36	24	0.23	0.20	970	29.8	3.5	0.27	2.2
	25-Jul	110	1.65	<0.36	NA	NA	0.29	770	58.8	11.1	0.36	6.8
	18-Aug	206	0.88	<0.18	NA	NA	0.16	1050	45.3	6.3	0.39	3.9
MEAN			1.35	<0.30	NA	NA	0.22	930	44.7	7.0	0.34	4.3

Table 13. Cont'd.

Location	Date	Anal#	Fe	Mn	TDN	TDP	NH4	DOC	SC	SN	SP	Chl-a
			µm/L									µg/L
NW	18-Jun	24	0.75	0.36	22	0.16	0.51	610	35.5	4.3	0.26	1.3
	24-Jul	107	1.10	<0.36	NA	NA	0.54	600	23.2	3.0	0.26	2.6
	17-Aug	195	0.44	<0.18	NA	NA	0.39	660	17.0	2.9	0.19	1.9
MEAN			0.77	<0.30	NA	NA	0.48	623	25.2	3.4	0.24	1.9
NE	18-Jun	25	1.13	<0.36	21	0.23	0.78	2030	45.3	5.9	0.37	2.3
	24-Jul	106	1.10	<0.36	NA	NA	0.60	590	27.7	3.5	0.30	4.1
	17-Aug	196	<0.36	<0.18	NA	NA	0.17	670	46.5	9.3	NA	3.4
MEAN			<0.06	<0.30	NA	NA	0.52	1097	39.8	6.2	0.34	3.3
F	18-Jun	26	1.13	<0.36	29	0.29	0.12	2260	47.5	5.3	0.53	3.3
	24-Jul	105	1.10	<0.36	NA	NA	0.46	870	83.3	14.4	0.70	11.1
	17-Aug	194	<0.36	<0.18	NA	NA	0.73	1050	128.7	24.6	0.99	23.2
MEAN			<0.86	<0.30	NA	NA	0.44	1393	86.5	14.7	0.74	12.5
EM	18-Jun	27	0.75	<0.36	30	0.13	0.24	2450	115.5	7.6	0.48	3.2
	23-Jul	101	0.55	<0.36	NA	NA	<0.03	1480	95.2	15.6	0.44	7.9
	17-Aug	197	<0.36	<0.18	NA	NA	0.12	1030	244.3	42.5	0.95	23.8
MEAN			<0.55	<0.30	NA	NA	<0.13	1653	151.7	21.9	0.62	11.6
CM	18-Jun	28	0.75	0.36	29	0.23	0.36	1030	44.8	5.8	0.54	2.6
	23-Jul	102	1.10	<0.36	NA	NA	0.42	1710	33.7	5.2	0.55	3.2
	17-Aug	198	<0.36	<0.18	NA	NA	0.84	990	58.7	9.3	0.37	7.3
MEAN			<0.74	<0.30	NA	NA	0.54	1243	45.7	6.8	0.49	4.4

Table 13. Cont'd.

Location	Date	Anal#	Fe	Mn	TDN	TDP	NH4	DOC	SC	SN	SP	Chl-a
							µm/L					µg/L
WM	18-Jun	29	0.75	0.36	29	0.35	0.66	720	44.0	5.6	0.54	< 0.1
	23-Jul	103	1.10	<0.36	NA	NA	0.97	1300	91.5	15.9	0.86	9.5
	17-Aug	199	<0.36	<0.18	NA	NA	0.46	1080	192.5	35.0	1.37	34.6
MEAN			<0.74	<0.30	NA	NA	0.70	1033	109.3	18.8	0.92	<14.7
I	19-Jun	34	3.01	0.36	44	0.77	0.20	3080	126.3	10.8	1.15	5.4
	25-Jul	112	9.91	1.93	NA	NA	1.78	870	136.3	29.6	1.50	26.0
	18-Aug	209	0.44	<0.18	NA	NA	9.27	1340	53.2	8.2	0.84	6.8
MEAN			4.45	<0.83	NA	NA	3.75	1763	105.3	16.2	1.16	12.7
WB	20-Jun	43	0.75	<0.36	22	0.16	0.49	680	24.5	2.3	0.28	1.1
	20-Jul	92	1.10	<0.36	21	0.19	0.50	600	39.0	4.9	0.28	3.1
	24-Aug	237	0.88	<0.18	23	0.29	0.54	690	35.3	4.9	0.21	2.2
MEAN			0.91	<0.30	22	0.22	0.51	657	32.9	4.0	0.26	2.1

Table 14. Other limnological variables at 1981 sampling sites; total suspended solids (TSS), temperature (TEMP), daily primary production of carbon (PROD), molecular oxygen concentration (O_2 CONC), relative saturation (O_2 SAT) and oxygen consumption in the water column (O_2 RESP), and absorbances at 455 nm after centrifugation and at 543 nm before filtration (UNFILT), after filtration (FILT), and after centrifugation (CENT) at 13 sites in the area of the Churchill River diversion, 1981. The "MEAN" primary production value is the mean integral daily carbon fixed by photosynthesis over the period 19 June to 18 August inclusive (61 days). The analytical number (Anal #) is an internal reference number.

Location	Date	Anal#	TSS mg/L	TEMP °C	PROD carbon $mg/m^{-2}d$	O_2			Absorbances at wavelengths 455 nm (ABS)	543 nm		
						CONC µm/L	SAT %	RESP µm/m^{-3}hr		UNFILT	FILT	CENT
G	19-Jun	35	6.1	NA	194	327	NA	107	0.034	0.082	0.022	0.011
	25-Jul	111	NA	17.78	823	290	97	294	0.031	0.115	0.021	0.010
	18-Aug	207	4.0	20.55	590	290	102	438	0.028	0.083	0.020	0.012
MEAN			5.0	19.17	587	302	100	280	0.031	0.093	0.021	0.011
6	19-Jun	33	5.9	14.09	122	354	109	247	0.034	0.201	0.032	0.013
	25-Jul	113	8.8	16.69	278	295	96	265	0.038	0.380	0.070	0.013
	18-Aug	208	5.3	18.90	284	281	96	218	0.033	0.183	0.051	0.012
MEAN			6.7	16.56	238	310	100	243	0.035	0.255	0.051	0.013
4	19-Jun	31	8.0	4.73	96	385	95	519	0.040	0.264	0.085	0.017
	25-Jul	109	5.3	NA	551	325	NA	408	0.040	0.162	0.058	0.015
	18-Aug	205	NA	17.36	228	303	100	290	0.036	0.177	0.042	0.015
MEAN			6.6	11.05	310	338	98	406	0.039	0.201	0.062	0.016
5	19-Jun	30	3.2	7.12	224	358	94	755	0.072	0.064	0.036	0.025
	25-Jul	108	3.3	16.55	499	304	99	219	0.065	0.110	0.038	0.022
	18-Aug	204	3.1	17.97	414	283	95	343	0.062	0.099	0.028	0.022
MEAN			3.2	13.88	408	315	96	439	0.066	0.091	0.034	0.023

Table 14. Cont'd.

Location	Date	Anal#	TSS mg/L	TEMP °C	PROD carbon mg/m^{-2}d	O_2			Absorbances at wavelengths 455 nm (ABS)	543 nm		
						CONC µm/L	SAT %	RESP µm/m^{-3}hr		UNFILT	FILT	CENT
SA	19-Jun	32	3.5	16.00	170	294	95	169	0.165	0.179	0.085	0.065
	25-Jul	110	2.1	NA	866	289	NA	545	0.108	0.110	0.050	0.039
	18-Aug	206	3.0	18.63	441	268	91	380	0.042	0.108	0.044	0.033
MEAN			2.9	17.32	543	284	93	365	0.105	0.132	0.060	0.046
NW	18-Jun	24	NA	10.36	140	332	95	142	0.043	0.068	NA	0.021
	24-Jul	107	NA	17.25	1177	275	91	267	0.038	0.075	0.020	0.010
	17-Aug	195	1.7	18.48	260	275	93	164	0.040	0.063	0.025	0.012
MEAN			NA	15.36	734	294	93	191	0.040	0.069	0.023	0.014
NE	18-Jun	25	3.3	8.74	199	344	95	354	0.045	0.127	NA	0.018
	24-Jul	106	NA	12.15	534	278	82	252	0.040	0.092	0.023	0.012
	17-Aug	196	2.1	18.44	640	293	99	412	0.035	0.065	0.015	0.010
MEAN			2.7	13.11	436	305	92	339	0.040	0.095	0.019	0.013
F	18-Jun	26	4.6	13.74	186	298	92	114	0.070	0.142	NA	0.022
	24-Jul	105	NA	17.80	1928	285	95	536	0.062	0.152	0.027	0.018
	17-Aug	194	5.8	18.63	1421	283	96	1003	0.063	0.120	0.026	0.018
MEAN			5.2	16.72	1262	289	94	551	0.065	0.138	0.027	0.019
EM	18-Jun	27	3.2	14.67	321	302	95	151	0.044	0.082	NA	0.014
	23-Jul	101	4.1	20.50	1462	302	106	523	0.038	0.088	0.018	0.012
	17-Aug	197	7.3	19.23	1682	303	104	1117	0.040	0.182	0.020	0.010
MEAN			4.9	18.13	1238	302	102	597	0.041	0.117	0.019	0.012

Table 14. Cont'd.

Location	Date	Anal#	TSS mg/L	TEMP °C	PROD carbon mg/m^{-2}d	O_2			Absorbances at wavelengths 455 nm (ABS)	543 nm		
						CONC µm/L	SAT %	RESP µm/m^{-3}hr		UNFILT	FILT	CENT
CM	18-Jun	28	2.7	12.19	215	305	91	73	0.060	0.091	NA	0.020
	23-Jul	102	1.6	19.06	281	284	97	247	0.051	0.060	0.024	0.016
	17-Aug	198	2.9	18.96	1055	297	101	538	0.050	0.082	0.020	0.013
MEAN			2.4	16.74	415	295	96	286	0.054	0.078	0.022	0.016
WM	18-Jun	29	2.0	11.34	171	251	73	122	0.055	0.059	NA	0.019
	23-Jul	103	3.7	18.21	1340	266	90	468	0.050	0.097	0.022	0.018
	17-Aug	199	7.7	19.33	1720	300	104	1295	0.054	0.175	0.025	0.014
MEAN			4.5	16.29	1113	272	89	628	0.053	0.110	0.024	0.017
I	19-Jun	34	3.3	14.59	205	224	70	NA	0.147	0.144	0.064	0.053
	25-Jul	112	NA	16.53	1523	248	81	727	0.190	0.178	0.068	0.062
	18-Aug	209	1.7	19.12	392	192	67	964	0.178	0.106	0.068	0.061
MEAN			2.5	16.75	972	221	73	896	0.172	0.143	0.067	0.059
WB	20-Jun	43	3.8	15.80	224	NA	NA	NA	0.055	0.123	0.040	0.021
	20-Jul	92	3.5	18.97	545	279	95	376	0.040	0.108	0.030	0.014
	24-Aug	237	NA	20.43	339	271	95	260	0.032	0.102	0.028	0.011
MEAN			3.6	18.40	451	275	95	318	0.042	0.111	0.033	0.015

Table 15. Concentration of total mercury ($\mu g \cdot g^{-1}$ dry weight) in various zooplankton size fractions from 13 sites in the area of the Churchill River diversion, 1981. See methods for parameter codes and sampling times.

Site*	Z73(1)	Z73(2)	Z73(3)	MZ73	Z153(1)	Z153(2)	Z153(3)	MZ153	Z351(1)	Z351(2)	Z351(3)	MZ351	Z760(1)	Z760(2)	Z760(3)	MZ760
CM	0.050	0.48	0.04	0.190	0.019	0.64	0.16	0.273	0.04	1.12	0.26	0.461	no sample	0.22	no sample	0.22
EM	0.039	1.65	0.39	0.693	0.007	0.86	0.27	0.374	0.046	5.00	1.32	2.122	no sample			
F	0.074	1.42	no sample	0.747	0.018	1.37	0.66	0.683	0.050	1.88	1.07	1.00	0.40	1.51	1.12	0.890
G	0.020	0.38	1.23	0.543	0.011	no sample	0.63	0.321	0.089	0.63	0.32	0.346	no sample	no sample	<0.07	<0.07
I	0.063	0.69	0.08	0.278	0.075	no sample	0.11	0.093	no sample	no sample	0.09	0.09	no sample	no sample	0.08	0.08
WB	0.034	3.45	no sample	1.742	0.009	3.19	no sample	1.600	0.0233	no sample	no sample	0.023	no sample			
NE	0.057	2.60	0.88	1.179	0.030	4.38	0.33	1.647	0.053	2.370	1.937	1.451	no sample	3.00	no sample	3.00
NW	0.10	3.20	0.84	1.38	0.19	4.56	1.08	1.943	no sample	3.20	1.67	2.435	no sample			
4	0.15	0.42	1.72	0.763	no sample	0.29	0.21	0.250	no sample	0.54	0.79	0.665	no sample			
5	0.021	0.72	0.28	0.340	0.021	1.66	0.26	0.647	no sample	1.54	0.46	1.00	no sample			
SA	0.050	1.24	0.35	0.547	0.054	1.96	no sample	1.007	0.007	0.12	8.74	2.956	0.009	0.20	3.24	1.150
6	0.10	0.69	0.03	0.273	no sample	2.90	0.03	1.465	no sample	3.93	0.28	2.105	no sample			
WM	0.11	1.41	0.28	0.60	0.058	no sample	1.19	0.624	0.021	2.69	0.40	1.037	no sample	no sample	0.20	0.20

* See Table 1 for key.

Table 16. Correlation coefficients between the mean concentrations of total mercury in yellow perch muscle (YPHG) and spottail shiner muscle (STHG) (Tables 8 and 9) and the concentration of total mercury of the upper 2 cm of unflooded soil (Table 1), of the upper horizons of unflooded soil (Table 2), and of the sediment (Table 4), the apparent depletion of total mercury in soil horizons (Tables 2 and 4), the means of various limnological parameters (Tables 12, 13, and 14), and the means of the concentrations of total mercury in various net plankton size fractions (Table 15) for 13 sites in the area of the Churchill River diversion, 1981. Number of data pairs for each correlation coefficient is shown in parentheses. See methods for parameter codes.

	YPHG	STHG
STHG	0.65 (6)	-
Upper 2 cm of soil Hg	-0.01 (7)	0.09 (12)
Moss/peat/litter Hg	-0.01 (7)	0.09 (12)
Ah Hg	0.72 (7)	0.64 (12)
Sediment Hg	0.42 (7)	-0.30 (12)
Moss/peat/litter Hg depletion	0.34 (3)	-0.16 (5)
Ah Hg depletion	0.74 (5)	0.63 (10)
Clay Hg depletion	0.29 (5)	0.02 (10)
Upper 2 cm of soil Hg depletion	0.34 (3)	-0.16 (5)
Mean water temp	-0.58 (7)	-0.84 (12)
Mean TDN	-	-
Mean TDP	-	-
Mean TSS	0.01 (7)	-0.27 (12)
Mean suspended P	0.13 (7)	-0.21 (12)
Mean suspended N	0.01 (7)	-0.25 (12)
Mean suspended C	-0.15 (7)	-0.32 (12)
M PROD	0.11 (7)	-0.21 (12)
M DOC	-0.24 (7)	-0.35 (12)
M SAT	-0.09 (7)	-0.35 (12)
M RESP	0.02 (7)	-0.10 (12)
M COND	0.69 (7)	0 (12)
M ABS	0.05 (7)	0.08 (12)
M ΔFILT	0.04 (7)	-0.19 (12)
M ΔCENT	-0.16 (7)	0.14 (12)
M Z73	0.03 (7)	-0.01 (12)
M Z153	0.57 (7)	0.26 (12)
M Z351	-0.03 (7)	0.24 (12)
M Z760	0.87 (5)	0.89 (6)

Table 17. Correlation coefficients between the mean concentrations of total mercury in yellow perch muscle (YPHG) and spottail shiner muscle (STHG) (Tables 8 and 9) and the values of various limnological parameters (Tables 12, 13, and 14) and the concentration of total mercury in various net plankton size fractions (Table 15) for 13 sites in the area of the Churchill River diversion, 1981. Number of data pairs for each correlation coefficient is shown in brackets. See methods for parameter codes.

	YPHG	STHG		YPHG	STHG
TEMP 1	-0.23 (6)	-0.78 (11)	SAT 1	0.21 (6)	0.18 (10)
TEMP 2	-0.19 (7)	-0.69 (10)	SAT 2	-0.64 (5)	-0.45 (8)
TEMP 3	-0.68 (7)	-0.85 (12)	SAT 3	0.01 (7)	-0.33 (12)
TDN 1	0.00 (7)	-0.26 (12)	RESP 1	0.08 (6)	0.68 (11)
TDN 2	-	-	RESP 2	-0.03 (7)	-0.37 (12)
TDN 3	-	-	RESP 3	0.11 (7)	-0.28 (12)
TDP 1	0.01 (7)	0.13 (12)	COND 1	0.49 (7)	-0.02 (12)
TDP 2	-	-	COND 2	0.86 (4)	0.26 (9)
TDP 3	-	-	COND 3	0.78 (7)	0.03 (12)
TSS 1	-0.13 (7)	0.04 (11)	ABS 1	0.14 (7)	-0.03 (12)
TSS 2	-0.68 (4)	0.14 (8)	ABS 2	0.01 (7)	0.09 (12)
TSS 3	0.09 (7)	-0.53 (10)	ABS 3	0.03 (7)	0.37 (12)
SP 1	0.03 (7)	-0.12 (12)	ΔFILT 1	-0.56 (3)	0.09 (6)
SP 2	0.11 (7)	-0.18 (12)	ΔFILT 2	0.12 (7)	-0.01 (12)
SP 3	0.18 (7)	-0.20 (11)	ΔFILT 3	-0.05 (7)	-0.26 (12)
SN 1	-0.12 (7)	0.16 (12)	ΔCENT 1	0.00 (3)	0.01 (6)
SN 2	0.03 (7)	-0.22 (12)	ΔCENT 2	0.08 (7)	0.17 (12)
SN 3	0.01 (7)	-0.30 (12)	ΔCENT 3	-0.19 (7)	-0.04 (12)
SC 1	-0.33 (7)	-0.09 (12)	Z73 (1)	0.47 (7)	0.35 (12)
SC 2	-0.07 (7)	-0.38 (12)	Z73 (2)	0.73 (7)	0.15 (12)
SC 3	-0.05 (7)	-0.34 (12)	Z73 (3)	-0.75 (6)	0.19 (10)
PROD 1	-0.42 (7)	-0.33 (12)	Z153 (1)	0.04 (7)	0.44 (10)
PROD 2	0.15 (7)	-0.09 (12)	Z153 (2)	0.35 (4)	0.26 (10)
PROD 3	0.24 (7)	-0.30 (12)	Z153 (3)	0.16 (7)	0.03 (10)
DOC 1	-0.31 (7)	-0.22 (12)	Z351 (1)	-0.53 (5)	-0.01 (8)
DOC 2	0.14 (7)	-0.33 (12)	Z351 (2)	-0.22 (6)	-0.05 (11)
DOC 3	-0.12 (7)	-0.31 (12)	Z351 (3)	0.12 (7)	-0.07 (11)
			Z760 (1)	-	-
			Z760 (2)	-	0.99 (4)
			Z760 (3)	0.91 (4)	0.35 (4)

Table 18. Correlation coefficients between the mean concentrations of total mercury in yellow perch muscle (YPHG) and spottail shiner muscle (STHG) (Tables 10 and 11) and the concentration of total mercury in upper soil horizons of unflooded soil (Table 3), and the apparent depletion of total mercury in upper soil horizons (Tables 3 and 6) for seven sites in the area of the Churchill River diversion, 1982. Number of data pairs for each correlation coefficient is shown in brackets.

	YPHG	STHG
Moss/peat/litter Hg	-0.12 (7)	-0.05 (7)
Ah Hg	-0.30 (7)	-0.19 (7)
Moss/peat/litter Hg depletion	0.24 (5)	0.17 (5)
Ah Hg depletion	0.33 (5)	0.48 (5)

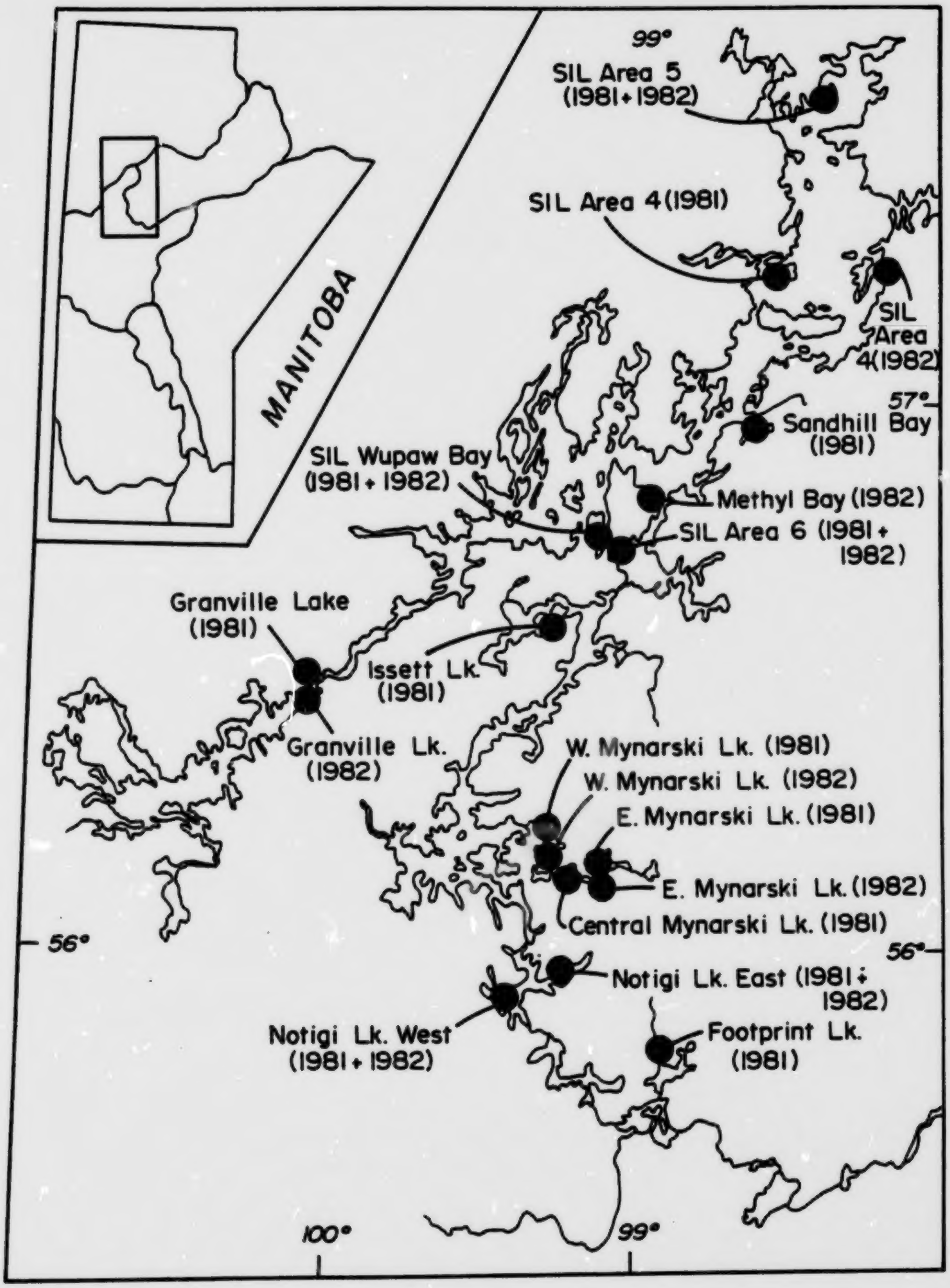
MANITOBA
99°
SIL Area 5 (1981+1982)
SIL Area 4(1981)
SIL Area 4(1982)
57°
Sandhill Bay (1981)
SIL Wupaw Bay (1981+1982)
Methyl Bay (1982)
SIL Area 6 (1981+ 1982)
Granville Lake (1981)
Issett Lk. (1981)
Granville Lk. (1982)
W. Mynarski Lk. (1981)
W. Mynarski Lk. (1982)
E. Mynarski Lk. (1981)
E. Mynarski Lk. (1982)
Central Mynarski Lk. (1981)
56°
56°
Notigi Lk. East (1981+ 1982)
Notigi Lk. West (1981+ 1982)
Footprint Lk. (1981)
100°
99°

Fig. 1. Map of study area showing sample sites.

Canadian Technical Report of
Fisheries and Aquatic Sciences 1490

March 1987

MERCURY METHYLATION AND DEMETHYLATION STUDIES AT SOUTHERN INDIAN LAKE, MANITOBA: 1981-1983

by

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This is the 3rd Technical Report
from the Central and Arctic Region, Winnipeg

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SPECIAL NOTE

This report was prepared for the Canada-Manitoba Agreement on the Study and Monitoring of Mercury in the Churchill River Diversion. Any conclusions, recommendations, or opinions expressed herein are those of the author and are not necessarily the same as those of the Steering Committee for this Agreement.

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Cat. no. Fs 97-6/1490E ISSN 0706-6457

Correct citation for this publication is:

Ramlal, P.S., C. Anema, A. Furutani, R.E. Hecky, and J.W.M. Rudd. 1987. Mercury methylation and demethylation studies at Southern Indian Lake, Manitoba: 1981-1983. Can. Tech. Rep. Fish. Aquat. Sci. 1490: v + 35 p.

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ABSTRACT

Ramlal, P.S., C. Anema, A. Furutani, R.E. Hecky, and J.W.M. Rudd. 1987. Mercury methylation and demethylation studies at Southern Indian Lake Manitoba: 1981-1983. Can. Tech. Rep. Fish. Aquat. Sci. 1490: v + 35 p.

From 1981 to 1983, studies were made on the processes of mercury methylation and demethylation at the Southern Indian Lake (SIL) reservoir. Specific rates of mercury methylation measured in the lake sediments during a survey of flooded lakes was generally related to the total mercury measured in fish muscle. Limnocorral studies demonstrated how an actively methylating microbial community becomes established after major environmental perturbations. These populations could become established under conditions where nutrients and mercury were readily available. In the limnocorrals used for more than one field season, the bacteria began to produce methyl mercury as soon as more mercury was added to the limnocorrals. A number of different materials, such as clay-mineral sediment and moss, were tested to determine their effect on mercury methylation. Moss in the short term could inhibit methylation, possibly by absorbing the added mercury. SIL bank material (clay-rich) bound less mercury than clay sediment from a mercury contaminated lake in Ontario, Wabigoon Lake, and much of the bound mercury could later be released.

Key words: mercury; methyl mercury; methylation; demethylation; reservoirs (water); limnocorrals.

RÉSUMÉ

Ramlal, P.S., C. Anema, A. Furutani, R.E. Hecky, and J.W.M. Rudd. 1987. Mercury methylation and demethylation studies at Southern Indian Lake Manitoba: 1981-1983. Can. Tech. Rep. Fish. Aquat. Sci. 1490: v + 35 p.

De 1981 à 1983, on a mené des études sur les processus de méthylation et de déméthylation du mercure dans le lac Southern Indian. Les taux spécifiques de méthylation du mercure tels que quantifiés dans les sédiments lacustres au cours d'un relevé des lacs inondés étaient généralement en relation avec le mercure total présent dans les muscles de poisson. À l'aide d'études en limnocorrals, on a montré comment une communauté microbienne en état de méthylation active s'établit après d'importantes perturbations environnementales. Ces populations peuvent s'établir dans des conditions où les bioéléments et le mercure sont facilement disponibles. Dans les limnocorrals utilisés pendant plus d'une saison de travaux sur le terrain, les bactéries ont produit du mercure méthylé dès l'apport de mercure. On a mis à l'essai différentes matières dont de la mousse et des sédiments argile-minéral afin de déterminer leur incidence sur la méthylation du mercure. À court terme, la mousse peut inhiber la méthylation par absorption possible due mercure ajouté. Les matières composant les rives du lac (riches en argile) ont retenu moins de mercure que les sédiments argileux d'un lac ontarien contaminé (lac Wabigoon); la plus grande partie du mercure lié peut être libérée ultérieurement.

Mots-clés: mercure; mercure méthylé; méthylation; déméthylation; réservoirs (de l'eau); limnocorrals.

INTRODUCTION

In 1976 Southern Indian Lake (SIL), in northern Manitoba, was impounded as part of the diversion of the Churchill River for hydroelectric development (Newbury et al. 1984). Fish muscle mercury levels increased in lakes flooded by the diversion and the observed increase of fish mercury levels was due to enhanced bacterial methylation of naturally occuring mercury in the flooded soils (Bodaly and Hecky 1979; Bodaly et al. 1984a).

Inorganic mercury and compounds containing mercury can be rapidly transformed by microbial action in aquatic environments. Methylated mercury is the most toxic of the compounds produced by these transformations (Brosset 1981). Environmental methyl mercury concentrations are regulated by the concurrent processes of production and degradation (e.g. Brosset 1981; Lexmond et al. 1976). Methyl mercury is biologically produced both by microorganisms which metabolize the mercuric (Hg^{2+}) ion to detoxify their environment (Pan-Hou and Imura 1982) and as a result of incorrect synthesis of organic molecules such as amino acids (Wood et al. 1972). Other bacteria then eliminate methyl mercury by converting it to methane and elemental mercury (Hg^0) (Tonomura and Kanzaki 1969; Tonomura et al. 1972). When inorganic mercury enters an aquatic ecosystem much of it is adsorbed on to organic and inorganic particles in the water (Benes and Havlik 1979; Rudd et al. 1980b). These organic particles (mostly phytoplankton and biological debris) can subsequently undergo decomposition, freeing previously bound inorganic mercury for biological methylation; sorption/desorption equilibria of mercury determine its distribution between dissolved and inorganic particle phases (Santschi et al. 1986). Mercury must be available as the dissolved mercuric ion for biological methylation to occur (DeSimone et al. 1973).

Methyl mercury is transferred through the food chain via a number of routes. It may adsorb onto, or be absorbed by, phytoplankton (Windom and Kendall 1979). It can also enter fish through the gills (Olson et al. 1973). Armstrong and Hamilton (1973) found that mercury concentrations of organisms in a mercury contaminated system was related to food selection. They found that omnivorous organisms, detritus feeders, and bottom dwelling invertebrates had considerably higher mercury levels than was found in either herbivorous organisms or those feeding on zooplankton. Little is known of the chemical form of mercury in invertebrates except that the percentage of total mercury methylated is quite low (Huckabee et al. 1979). Methyl mercury can also be produced by the intestinal contents of fish and bioaccumulation from the intestine can occur (Rudd et al. 1980a).

In 1981, the following studies on the microbiological transformations of mercury at SIL began. They are described under three headings:

1: Limnocorrals
   Sediment was collected from limnocorrals subjected to various treatments to find out how quickly an actively methylating population of bacteria could be established, and what conditions were conducive to enhanced mercury methylation.

2: Survey Studies
   Specific rates of methylation and demethylation of mercury were measured in sediment and water samples collected from flooded shorelines and offshore "old lake" sediment in and around SIL.

3: Additional Studies
   a. The capacity of natural materials such as lake sediment and lake shore soils to fix or adsorb mercury was tested by examining the effect of these materials on mercury methylation.

   b. Bodaly et al. (1984) found that levels of mercury in the upper organic soil horizon were much higher than inorganic subsoil layers. We have conducted an experiment to find out how erosion of this upper organic layer by flooding would affect the availability of mercury for methylation.

1: LIMNOCORRALS

METHODS AND MATERIALS

Site description

Beginning in 1981, limnocorrals were set up in a small bay off of Wupaw Bay (previously known as Long Bay) in SIL (Fig. 1.1; Hecky et al. 1987). Detailed explanation of the limnocorral experiments is available in Hecky et al. (1987). The treatments added to the limnocorrals are listed in Table 1.1. At various times during July and August of 1981, 1982, and 1983, water and sediment were collected from selected limnocorrals and the specific rates of mercury methylation and demethylation determined.

Sample collection and manipulation

Until August 23, 1981, sediment was collected using a modified Ekman dredge (Burton and Flannagan 1973). Flocculent material at the sediment water interface (0-2 cm) was siphoned with a hand vacuum pump into 500 mL PVC bottles. Water samples were obtained 30 cm above the undisturbed sediment surface. Limnocorral sediments of August 23, 1981, and all samples until August 16, 1982, were taken with a submersible pump and collected in a 20 L carboy. The suspended material was allowed to settle overnight, the supernatant was decanted and the settled 'floc' resuspended in 500 mL of lake water.

After August 16, 1982, samples were collected from each site by a submersible electric pump the intake of which was dragged over the sediment surface and the fluid pumped into 2 L or 20 L carboys. The sediment was treated as described above before methylation measurements were done.

In 1983, the samples were collected using the same apparatus described in 1982, but the surface sediment was siphoned into 2 L PVC bottles and the supernatant was not decanted.

Radioisotope addition and sample incubation

In the laboratory, the sediment suspension was mixed and 100 mL was measured into each of three acid washed 125 mL glass reagent bottles, one as a control and two as replicate tests. When demethylation was included (in 1983) an additional three subsamples were taken. The microbial activity of the sediment controls was stopped with 1 mL of 4N HCl prior to isotope addition. Sediments collected from anoxic sites were incubated with nitrogen in the head space of the incubation vessels to maintain anaerobic conditions. All other samples were incubated with room air in the headspace. Methylation measurements of water was done using the above procedure, with 100 mL of water.

Specific rate of mercury methylation: Methylation samples were injected with ^{203}Hg as $HgCl_2$ at an approximate dosage of 74 kBq·2 µg Hg^{-1}·g dry $sediment^{-1}$. The samples were shaken for 1 min after isotope addition then incubated at *in situ* temperatures for 12 h. Following incubation, 1 mL of 4N HCl was added to each sample and the alkylated ^{203}Hg was extracted using an organic solvent extraction technique (Furutani and Rudd 1980). The ^{203}Hg activity of the samples was determined with a liquid scintillation counter (Packard 3033).

Specific rate of demethylation of methyl mercury: In 1983, the rate of microbial breakdown of methyl mercury, or demethylation, was determined by adding ^{14}C-methylmercuric iodide to a set of samples separate from those used for the methylation studies. The isotope was added at a dose of 14.8 kBq·0.2 µg Hg (as $CH_3HgI)^{-1}$·g dry $sediment^{-1}$. The samples were shaken for 1 min after isotope addition and incubated for 12 h at the *in situ* temperatures. After incubation, the samples were acidified and the volatile ^{14}C and products of demethylation were stripped from the samples (Ramlal et al. 1986). The ^{14}C activity of the samples was determined with a liquid scintillation counter (Packard 3033).

Calculation of specific rate of biological mercury methylation and degradation: The following equations were used to determine the specific rate of microbial mercury methylation (M) after the radioactive counts had been quench corrected. Methyl mercury degradation (D) is calculated using the same equations.

1. Total µg Hg methylated =

$$\frac{(\text{dpm}_{\text{sample}} - \text{dpm}_{\text{blank}})\cdot\text{scale factor}}{\text{added dpm}\cdot\mu\text{g}^{-1}\ \text{Hg}}$$

2. ng Hg methylated·g dry $sediment^{-1}\cdot h^{-1}$ =

$$\frac{\text{Total }\mu\text{g Hg methylated}\cdot(1000\ \text{ng}\cdot\mu\text{g}^{-1})}{\text{dry weight (g)}\cdot\text{incubation time (h)}}$$

3. % of added Hg methylated·g dry $sediment^{-1}\cdot h^{-1}$ =

$$\frac{[\text{ng Hg methylated}\cdot\text{g dry sediment}^{-1}\cdot\text{h}^{-1}]\cdot 100}{\text{added ng Hg}}$$

Microbial activity: In 1982, microbial metabolic activity was measured by monitoring the oxygen consumption of the unconcentrated sediment/water mixtures using Winkler titrations (Stainton et al. 1977).

In 1983, microbial activity of the sediments was monitored by measuring the rate of production of carbon dioxide. Replicate samples of sediment were added to test tubes of a known volume, with either room air or helium (for sediments which had no oxygen in overlying water) in the head space. After incubation at the *in situ* temperatures for 12 h the sediment was acidified with HCl and the gas in the head space was collected for CO_2 determination by gas chromatography. Carbon dioxide was measured at the beginning and end of the incubation period. Measurement of methane production was attempted during one series of methylation/demethylation measurements, but methane could not be detected using a thermal conductivity gas chromatograph.

Chemical analyses: Subsamples of the sediment collected were taken for total suspended solids, carbon and nitrogen analyses (Stainton et al. 1977). The pH of the sediment/water suspension was measured prior to sediment incubation and dry weights of material in suspension were determined. The results of the chemical analyses of the sediment are listed in the appendix.

RESULTS AND DISCUSSION

The results of some of the mercury in fish muscle are included in this section. These data are more fully described in Hecky et al. (1987) and are included here for comparative purposes (Figs. 1.2A-B).

Specific mercury methylation rates were measured on two dates in the limnocorrals in 1981 (Table 1.2). Mercury methylation was undetectable in the water column, was barely detectable in the phosphorus and nitrogen (P and N) limnocorral, but could be measured in the moss limnocorral (0.086 % methylated·g dry $sediment^{-1}\cdot h^{-1}$). The rate measured in the P and N limnocorral was close to those measured in sediment collected outside of the limnocorrals (Bay samples in Table 1.2). Mercury methylation was undetectable in both the high clay and control limnocorrals. Yellow perch collected from the moss enclosure had the highest measured total Hg concentrations in muscle tissue (Fig. 1.2A).

In 1982, the first measured methylation rates (July 8 in Table 1.3; Fig. 1.3) were higher in both the one year old control (0.008% methylated·g dry $sediment^{-1}\cdot h^{-1}$) and the enclosure containing the previous year's moss treatment (Table 1.1). Low methylation rates were measured in the limnocorral with bank clay addition on all of the dates it was sampled except August 23, but it still had the lowest rate measured of all the limnocorrals sampled on that date. These results are consistent with the field and laboratory studies (Section 2, Table 2.1 - South Bay and Table 2.2; Section 3, Table 3.3), in which no methylation was detected in bank clay or in clay-rich sediment. The sediment of the double moss addition had increasing specific methylation rates during the sampling period, and had the highest measured rate (0.132 % added Hg methylated·g dry sediment$^{-1}\cdot$h^{-1}) of all the limnocorrals.

There was no apparent relationship between the total Hg in fish muscle (Fig. 1.3A) and the mean specific mercury methylation rates (Fig. 1.3B). The fish of the double moss addition, the bank clay addition and the one year old moss limnocorral had high mercury levels, while mean methylation rates were high in moss limnocorrals (0.032-0.054% methylated·g dry sediment$^{-1}\cdot$h^{-1}) but low in the bank clay limnocorral (0.004% methylated·g dry sediment$^{-1}\cdot$h^{-1}).

In 1983, specific mercury methylation rates of sediment were consistently higher in re-used limnocorrals, regardless of the treatment (Tables 1.5A-B). Degradation of methyl mercury occurred in substantial amount in all limnocorrals regardless of age or treatment. A ratio of the specific methylation rate to the specific demethylation rate (M/D) was calculated (Table 1.5B). There did not seem to be any seasonal trends of M/D in the limnocorral sediment. The mean M/D was high in all re-used limnocorrals and low in the new limnocorrals which was primarily due to the differences in the mean methylation rates. Total Hg was highest in the fish of the control limnocorral (Fig. 1.4A), although uptake of ^{203}Hg was not exceptional (Hecky et al. 1987).

The limnocorral experiments show how bacterial populations can be altered when they are exposed to environmental perturbations such as additions of mercury. When limnocorrals were used for more than one year in the same site (Table 1.4) an actively methylating population of bacteria was already established. In all the limnocorrals used more than once, the initial methylation measurements the following year were higher than in new limnocorrals (Table 1.4). In 1983, the mean M/D ratio for the season for all reused limnocorrals was at least three times higher than in new limnocorrals regardless of the treatments they were given. In the control limnocorral which had received a substantial dose of stable mercury with each radioactive spike for two years, the mean specific methylation rate was 0.069% added Hg methylated·g dry sediment$^{-1}\cdot$h^{-1} and the fish mercury levels were the highest recorded for the three years of the limnocorral experiment (Fig. 1.4A and B). It seems likely that the sediment of this limnocorral had a high available mercury concentration after two mercury additions, and also a well established population of mercury methylating bacteria.

Table 1.1. Treatments used in the enclosures at SIL during 1981 to 1983. Each year, approximately 2 mg Hg was added to each limnocorral with radioisotope spike with the exception of limnocorral 2 in 1982. This increased the background mercury concentration in water by 5 to 10.

Limnocorral number	1981	1982	1983
1	control (n)	6 year-old moss undisturbed (n)	7 year-old moss (1) (flooded) disturbed
2	moss (n)	window screen (n)	spruce boughs (2)
3	P and N (n)	1x moss (n)	1x moss (n)
4	low clay (n)	2x moss (n)	control (6)
5	control (n)	1 year old control (1)	1982 clay addition (1)
6	high clay (n)	new control (n)	Two year old moss (8)
7	control (n)	bank clay (n)	1x moss (n)
8	bay	1 year old moss (2)	1x moss and lake clay addition (n)

(n) denotes new limnocorral; (1) denotes limnocorral number of previous year. In 1981, limnocorral number 8 was not a limnocorral, but the water sampled outside of the limncorrals.

Table 1.2. ^{203}Hg-methyl mercury production, SIL 1981 (as measured by % added mercury methylated·g dry sediment^{-1}·h^{-1}). Bay refer to samples taken near but outside limnocorrals.

Limnocorral	July 20	Aug. 3
moss	0.000 (w)	0.086
P & N		0.001
control		0.000
high clay	0.000 (w)	0.000
bay	0.002	0.001
bay	0.000 (w)	

(w) water samples

Table 1.3. ^{203}Hg-methyl mercury production at SIL 1982, limnocorrals (as measured by % of added mercury methylated·g dry sediment^{-1}·h^{-1}).

Limnocorral	July 8	July 21	Aug. 3	Aug. 23
6 yr moss	0.001	0.002	0.050	0.024
1x moss	0.002	0.001	0.023	0.020
2 x moss	0.005	0.005	0.072	0.132
1 yr control	0.008	0.007	0.000	0.012
control	0.003	0.002	0.001	0.015
bank clay	0.001	0.000	0.001	0.012
1 yr moss	0.012	0.004	0.002	0.109

Table 1.4. Comparison of early July (initial) mercury methylation rates for limnocorrals used more than one year (as measured by % added mercury methylated$\cdot$g dry sediment$^{-1}\cdot$h^{-1}).

Limnocorral	Years	Methylation Rate Year One	 Year Two
6-7 yr moss	1982-83	0.001	0.025
control	1982-83	0.003	0.049
clay	1982-83	0.001	0.025
moss	1982-83	0.012	0.082

Table 1.5A: Specific rates of biological methylation and demethylation of mercury by sediments collected from limnocorrals at SIL, 1983 (as measured by % of added mercury methylated or demethylated•g dry $\text{sediment}^{-1}\cdot h^{-1}$).

Limnocorral	July 10, 1983		July 25, 1983		August 7, 1983	
	D	M	D	M	D	M
7 yr moss disturbed	0.099	0.025	0.153	0.065	0.228	0.032
spruce	0.041	0.039	0.230	0.069	0.306	0.057
1x moss*	0.241	0.006	0.141	0.006	0.339	0.001
control	0.159	0.049	0.180	0.128	0.310	0.029
1982 bank clay	0.259	0.025	0.372	0.086	0.206	0.049
2 yr moss	0.148	0.082	0.209	0.059	0.188	0.028
1x moss*	0.113	0.001	0.352	0.006	0.291	0.007
1x moss & lake clay*	0.065	0.001	0.174	0.022	0.286	0.018

* These limnocorrals were new in 1983 (see Table 1.1).

Table 1.5B: Methylation/demethylation ratios (M/D) and average ratios of limnocorral sediment studies of 1983.

Limnocorral	July 10	July 25	August 7	Mean for all dates
7 yr moss disturbed	0.254	0.422	0.139	0.272
spruce	0.930	0.300	0.187	0.472
1x moss*	0.024	0.043	0.003	0.023
control	0.308	0.709	0.093	0.370
1982 bank clay	0.095	0.231	0.236	0.187
2 yr moss	0.552	0.284	0.146	0.327
1x moss*	0.011	0.016	0.025	0.017
1x moss & lake clay*	0.012	0.126	0.060	0.066

* These limnocorrals were new in 1983 (see Table 1.1).

2: SURVEY SAMPLES

METHODS AND MATERIALS

In 1981, sediment was sampled from South Bay on SIL, an area of severe bank erosion and high sedimentation (Hecky and McCullough 1984; Newbury and McCullough 1984). Sediment collected from Granville Lake, an unflooded lake upstream on the Churchill River from SIL, was used as control sediment. Sediment was obtained from Issett Lake, a small lake off the Churchill diversion which had extensive flooding and low flushing. Sediment was also collected from Methyl Bay, a relatively isolated bay on SIL. Methyl Bay is relatively shallow with slumping shorelines caused by melting permafrost. The bathymetry is unrecorded but depths >8 m occur. Sediment from this bay was taken from sites which had been inundated when the lake was impounded in 1976 as well as from sites on the original lake bed.

In 1982, in addition to sediment from Methyl Bay, Granville and Issett Lakes, samples were taken from East and West Mynarski, and West Notigi as well as Areas 4 and 5 in SIL (Fig. 1.1; Newbury et al. 1984). Unflooded E. Mynarski Lake was a control site, Areas 4 and 5 of SIL and West Notigi Lake were flooded (Bodaly et al. 1984b) but Area 4 had the least areal extent of flooding of any of surveyed sites i.e. lowest area flooded as a ratio of lake area after flooding (Newbury et al. 1984).

In 1983 survey samples were collected only from Methyl Bay.

Sample collection and all of the analyses were done as described in Section 1.

RESULTS AND DISCUSSION

In 1981, methylation was measured at three sites on two different dates (Table 2.1). The highest rates were measured in sediment collected from the flooded Issett Lake. Methylation was close to zero in sediments collected from both flooded South Bay and unflooded Granville Lake. Comparison of fish muscle mercury levels and methylation measured during the 1981 mercury survey were too limited to allow conclusions to be reached and the data is not included. Methylation was measured at only three sites and there were no complete collections of yellow perch or spottail shiners for each of these three.

Five sites were studied at Methyl Bay during 1982 (Table 2.2) where methylation was only detectable in sites which had been inundated by flooding. In the clay-rich offshore sediments, at greater depths than inshore sites but similar temperatures, there was virtually no methylation. There was a strong seasonal trend at the inundated sites with maximum methylation rates occurring in early to mid-August.

Mercury methylation rates measured on sediment collected during the August 17-18, 1982 survey (Table 2.3), show that the sediment from Issett, as in the previous year, had high rates (0.312% added $Hg \cdot g^{-1} \cdot h^{-1}$) relative to controls (i.e. Granville in 1981, Granville and East Mynarski in 1982; 0-0.015% added $Hg \cdot g^{-1} \cdot h^{-1}$), W. Mynarski sediment had a similar rate of 0.261% added $Hg \cdot g^{-1} \cdot h^{-1}$. The level of mercury methylation in W. Notigi 2.25% added $Hg \cdot g^{-1} \cdot h^{-1}$ was 10 times higher than that measured at these other inundated sites.

Sediment methylation rates and total mercury in fish muscle (Bodaly et al.1987; Figs. 2.1A and B), followed similar trends. Methylation was highest in W. Notigi sediment which had the highest mercury concentration in fish muscle. Fish mercury concentration and sediment methylation rates were also comparable in W. Mynarski and Area 5, SIL samples. Of the sites which exhibited the lowest methylation rate (unflooded control sites E. Mynarski and Granville, and flooded site in Area 4 in SIL) only fish collected from Area 4 had relatively high mercury levels. It is possible that the methylating bacteria at this site were sampled during a period of decline, or that the site sampled for methylation measurement was not representative of the habitat frequented by the fish.

In 1983, methylation was almost undetectable in the offshore clay rich sites of Methyl Bay (Table 2.4), flooded inshore sites had methylation rates ranging from 0.020-0.267% added $Hg \cdot g^{-1} \cdot h^{-1}$. A comparison of the M/D of the flooded and unflooded areas within the bay show significant differences. The M/D ratios of the flooded areas were as high as 0.824. It would appear that the flooding of Methyl Bay caused a shift in the M/D ratio so that the methylating population created a higher rate of net methyl mercury production than that in the unflooded sites.

Table 2.1. ^{203}Hg-methyl mercury production, SIL 1981 (as measured by % added mercury methylated$\cdot$g dry sediment$^{-1}\cdot$h^{-1} or % added mercury methylated$\cdot$L$^{-1}\cdot$h^{-1}).

Site	July 25	Aug. 18
Granville	0.000	0.000
South Bay	0.000	0.000
Issett	0.035	0.055
Granville (w)	0.000	0.000
South Bay (w)	0.000	0.000
Issett (w)	0.000	0.000

(w) water samples

Table 2.2. ^{203}Hg-methyl mercury production; Methyl Bay, SIL 1982 (as measured by % of added mercury methylated·g dry sediment^{-1}·h^{-1}).

Site	Depth (m)	July 6	July 12	July 19	Aug. 2	Aug. 16
clay	5.0	0.000	0.000	0.000	0.000	0.000
clay	5.0		0.000			
clay	5.0	0.000	0.000	0.000	0.000	0.001
flooded	2.5	0.000	0.005	0.008	0.077	0.093
flooded	3.0	0.001	0.000	0.007	0.126	0.021

Table 2.3. Hg-methyl mercury production of 1982 SIL surveys (as measured by by % of added mercury methylated·g dry sediment^{-1}·h^{-1}).

Site	Aug. 17
Mynarski E.	0.015
Mynarski W.	0.261
Notigi W.	2.251
Area 5, SIL	0.057
Area 4, SIL	0.011
Granville	0.004
Issett	0.312

Table 2.4. Specific rates of biological methylation and demethylation of mercury and the ratio of these processes (M/D) by sediments collected from three stations at Methyl Bay, SIL 1983 (as measured by % added mercury methylated or demethylated·g dry sediment^{-1}·h^{-1}).

Station	Depth (m)	July 19, 1983			July 19, 1983		
		D	M	M/D	D	M	M/D
clay	5.2	0.023	0.003	0.110	0.067	0.001	0.009
flooded	2.6	0.205	0.149	0.730	0.267	0.220	0.824
flooded	1.7	0.134	0.020	0.147	0.184	0.025	0.137

3: ADDITIONAL STUDIES

METHODS AND MATERIALS

Samples

Materials from the shoreline and backshores of SIL were used in the following laboratory experiments. These materials included clay sediment, clay soils, humic soils, cores from a Sphagnum moss bed and surface lake water (Table 3.1). Special care was used with the material required for the erosion experiment to ensure they were undisturbed cores or large intact pieces (e.g. clay sediment) from which cores were taken. Sediment was also collected from Wabigoon Lake in Ontario. This lake has been mercury contaminated and the sediment was used as a comparison to the sediments collected from SIL.

Inhibition of mercury methylation by various materials

Various materials were tested for their ability to inhibit methylation (Table 3.1). Distilled water was used as a control i.e. sample with no ability to adsorb mercury. Acid killed blanks (no microbial methylation occurring) contained 50 mL of distilled water and 1 mL of 4N HCl. Each of the materials was added to 125 mL glass reagent bottles and autoclaved for 20 minutes at 121°C. Fify mL of well mixed sediment from Lake 114 (Experimental Lakes Area) was added to the reagent bottles and served as both an inoculum of mercury methylating bacteria and their required nutrients. The sediment from Lake 114 is not contaminated by mercury and has been previously used as a source of mercury methylating bacteria (Furutani and Rudd 1980). The sediment consisted mainly of flocculent organic material. The amount of Lake 114 sediment used was more than enough required to overwhelm any differences in the nutrient composition between the various materials tested.

Samples to be incubated anaerobically were flushed with nitrogen for 5 min. During flushing the Lake 114 sediment and $^{203}HgCl_2$ was added to each bottle. Aerobically incubated samples had room air in the headspace of the incubation vessels. The aerobic samples were incubated with, and without agitation (150 rpm) during incubation. Samples were incubated for 24 hours.

Moss basket experiment

On July 25, 1982 an experiment testing the effectiveness of moss binding and/or scavenging mercury from the water began. This experiment was done in Wupaw Bay and Wupaw Bay bog at SIL, lasting from July 25, 1982 until August 26, 1983. Bundles of moss were suspended in the water column of each site to find out whether moss would pick up mercury from the water, and if so, whether some of this mercury would be converted to methyl mercury and sequestered in the moss. At various times (Table 3.2) subsamples were taken from the moss bundles at each site and frozen for storage and subsequent analysis of total and methyl mercury.

Total mercury was measured by atomic absorption and methyl mercury was estimated using a gas chromatograph equipped with an electon capture detector (Furutani and Rudd 1980; Uthe et al. 1972). To ensure that the methyl mercury was not overestimated, silver sulphate removal of the extracted methyl mercury was also done. The samples were then re-injected into the gas chromatograph (with an electron capture detector) and the peak height of material with the same retention time as methyl mercury was subtracted from the original measurement.

Comparison of distribution of ^{203}Hg between SIL bank material/water and Wabigoon Lake/sediment water

Decreasing amounts of SIL clay-rich bank material and Wabigoon Lake sediment were dispensed into 125 mL Erlenmeyer flasks. Distilled water (10 mL) was added to each of the flasks to final as concentration water:sediment of 2:1, 6:1, 25:1, and 50:1 for the SIL bank material and 11:1, 24:1, 32:1, and 90:1 for the Wabigoon Lake sediment suspensions. Radioactive mercuric chloride was added to each of the flasks (7500 dpm/flask) which were shaken for 2 min. Subsamples were removed, centrifuged at 6000 G for 30 min, and 2 mL of the supernatant was counted in a liquid scintillation counter.

The sediment pellets from the centrifuge tubes were resuspended in water at the same concentrations and left to stand for one week. Subsamples were again centrifuged, the supernatant counted and the % of the previously adsorbed Hg that was released after one week was calculated. The distribution ratio, K_d, (Nyffeler et al. 1984) was calculated for the ^{203}Hg:

$$K_d = \frac{C_s}{C_w}$$

where C_s is the concentration of ^{203}Hg in sediment ($dpm \cdot g^{-1}$) and C_w is the concentration of ^{203}Hg in water ($dpm \cdot mL^{-1}$).

The effect of soil perturbation by erosion on mercury methylation

Mercury methylation rates were measured on a variety of materials using 20-30 g wet weight (5 g dry weight) for sediments and soils, or 50 mL of lake water. Methylation was detectable only in the clay sediment sample (Table 3.3). Therefore, the "erosion" experiment was conducted with these sediment samples, and included humic and clay soil samples for comparison. Six cores were taken from a large intact piece of clay sediment. The cores, 3 cm in diameter and 5-6 cm deep were transferred to plexiglass jars of the same diameter. Four mL of sterile, distilled water was added to each of the soil samples to rehydrate them. Duplicate or triplicate cores of each type of sample were mixed with a glass rod to simulate erosion which would destroy the bacterial micro-environments. A total of 1 mL of $^{203}HgCl_2$ in 100 µL aliquots was injected into 10 different sites within each core. A 100 µL Hamilton syringe fitted with a fine needle was used for the injections into both the mixed and intact cores. One mL of 4 N hydrochloric acid was added to the $^{203}HgCl_2$ when the blank cores were injected. Following the 24 hour incubation the alkylated ^{203}Hg was extracted from the entire core (Furutani and Rudd 1980).

RESULTS AND DISCUSSION

Inhibition of mercury methylation of natural material

The results of this experiment (Table 3.1) have been interpreted with the assumption that high methylation rates are indicative of high dissolved (bio-available) mercury concentrations. This is a valid assumption since it has been demonstrated that mercury methylation is directly related to mercury concentration over several orders of magnitude 0.04-10 µg Hg$\cdot$g^{-1} sediment (Rudd et al. 1983). Distilled water with Lake 114 sediment should have the most bio-available mercury (as $^{203}HgCl_2$) because it had the highest methylation rate. The methylation rates and the corresponding bio-available mercury levels are expressed as a percentage of the rate found in the control. The comparable "% unavailable mercury" is the difference between the "% bio-available mercury" and "100% bio-available mercury" (Table 3.1).

There was virtually no mercury methylation detected in samples which contained room air in the headspace of the incubation vessel and which were shaken during incubation. In the samples containing room air, but not shaken during incubation, the methylation rate was much lower than the control (Table 3.1). These results indicated that atmospheric oxygen would inhibit mercury methylation in these samples. The unshaken, aerobic sample still methylated at a reduced rate because without agitation anaerobic micro-environments were probably present in the bottom of the sample during incubation.

Moss basket experiment

Total mercury and methyl mercury analyses were made of each of the moss-peat subsamples. The results (Table 3.2) do not show any evidence of significant methyl mercury accumulation or loss by the moss-peat material on a dry weight basis from the beginning to the end of the experiment.

Comparison of distribution of ^{203}Hg between SIL bank material/water and Wabigoon Lake sediment/water

The results of addition of ^{203}Hg to SIL bank material and Wabigoon Lake sediment (Table 3.4) show there are considerable differences between the two materials. The SIL bank material was clay-rich, and the Wabigoon Lake sediment had more sand than clay. The SIL bank material had considerably lower K_d than the Wabigoon Lake sediment, that is, there was more ^{203}Hg in the supernatant. After standing for one week, more ^{203}Hg had desorbed from the SIL material and although the K_d values were higher, they were still less than that of the Wabigoon material. It is unlikely that equilibrium was established after 32 min, but enough ^{203}Hg had been adsorbed to determine the K_d one week later.

The effect of soil perturbation by erosion on mercury methylation

The highest methylation rates measured in the erosion experiment were found in clay mineral sediment cores; low rates were found in the bank soils. When sediment cores were mixed, the methylation rates of both the clay sediment and bank soil increased three-fold (Table 3.3). Mercury methylation was not detected in either mixed or unmixed cores of the clay soil.

The total and methyl mercury concentrations of the three cores were measured (Table 3.3) and the methyl mercury results agree with the methylation results. The sediment sample had both the highest methylation rate and the most methyl mercury (Table 3.3). Both the humic soil and the sediment clay had 120 ng total Hg$\cdot$g dry sediment^{-1}, yet methyl mercury was undetectable in the humic soil.

Table 3.1. Effect of various materials on the bio-available mercury, SIL 1981.

Autoclaved Material	Incubation Headspace Shaken	Gas	MeHg (dpm/ sample)	Bio-availability Hg relative to distilled water	% Bound
control 50 mL dist. water	no	N_2	5638	100	0
clay 1.5g dry wt	no	N_2	3882	69	31
SIL unfilt. water 50 mL	no	N_2	3485	62	38
Sphagnum moss 1.5g dry wt	no	N_2	1075	19	81
poplar forest soil 1.5 g dry weight	no	N_2	3082	53	47
blank 50 mL acidified d water	no	N_2	27		
50 mL d water	no	air	428		
50 mL d water	yes	air	28		

* 50 mL ELA (L114) sediment (approx. 1.5 g dry weight) were added to all samples as bacterial inoculum and nutrient source.

Table 3.2. Methyl mercury and total mercury concentration of moss samples suspended in Wupaw Bay and Wupaw Bay bog, SIL.

Site	Date	Total mercury ($\mu g \cdot g^{-1}$)	Methyl mercury ($ng \cdot g^{-1}$)
Initial samples	July 15, 1982	0.08	7
		0.09	3
		0.07	4
		0.06	6
Lake	August 10, 1982	0.08	9
		0.08	<1
	October 18, 1982	0.09	<1
		0.11	<1
		0.07	<1
	August 26, 1983	0.06	3
		0.07	6
Bog	August 10, 1982	0.12	3
		0.09	8
	August 26, 1983	0.08	3
		0.08	<1

Table 3.3. Description of samples, water content, total mercury and methyl mercury concentration (per gram dry weight) and ^{203}Hg methyl mercury production in samples used for erosion experiments, SIL 1981.

Sample type	Description	% Water	Total Hg $ng \cdot g^{-1}$	MeHg $ng \cdot g^{-1}$	Sample Cond.	Methylation (dpm/sample)
Clay sediment	-fine roots and fibers throughout the core -dark brown to black with grey clay sections	82	120	1.8	unmixed	41
						47
					mixed	145
						112
						171
Humic soil	-roots and fibers throughout the core -organic texture -dark brown	71	120	UD	unmixed	1
						3
					mixed	5
						11
Clay soil	-no roots or fibers -light brown, grey clay	25	30	UD	unmixed	0
						0
					mixed	0

UD - undectectable-less than 0.3 $ng \cdot g^{-1}$ dry weight.

Table 3.4. Comparison of SIL bank material and Wabigoon Lake sediment ^{203}Hg binding and distribution ratios (K_d).

Lake	Dil'n*	dmp•mL^{-1}	dpm•g^{-1}	K_d	% bound ^{203}Hg
SIL (32 min)	2:1	34.0	1502	44.2	97.5
	6:1	39.2	4296	109.6	95.6
	25:1	304.0	11115	36.6	59.5
	50:1	272.4	23759	87.2	63.7
SIL (1 week)	13:1	26.4	9361	345.6	96.8
	25:1	65.7	9506	144.7	82.7
	50:1	79.4	19808	249.5	80.1
Wabigoon (32 min)	11:1	37.1	7160	193.0	94.4
	24:1	60.7	14811	243.9	90.9
	31:1	45.3	19460	429.6	93.2
	91:1	96.2	51986	540.4	85.5
Wabigoon (1 week)	24:1	33.8	11244	332.7	94.4
	31:1	29.6	14828	500.9	95.2
	91:1	38.2	38121	997.9	93.3

* Dilutions of the SIL bank material and Wabigoon Lake sediment were made using 10 mL of distilled water and decreasing amounts of sediment.

GENERAL DISCUSSION

The radiochemical methods we used (Furutani and Rudd 1980; Ramlal et al. 1986) measure the specific rates of methylation and demethylation. The radioactive mercury, both $^{203}Hg^{2+}$ and $^{14}CH_3Hg^+$, was added to samples in sufficient quantities to overwhelm the concentrations of these mercury species which might be present in the samples and are not conventional radiotracer methods. These methods are, however, more sensitive than those methods which use stable (cold) mercury to estimate both mercury methylation and demethylation. For detectable results the methods using stable mercury require long incubation periods and the addition of mercury at concentrations far exceeding background concentrations (and well above concentrations used in our assays; Billen et al. 1974; Spangler et al. 1973) of even the most contaminated sites (Armstrong and Hamilton 1973; Jackson and Woychuk 1980). The long incubation times and high mercury concentrations may cause the microbial populations to change from their natural state. In contrast, it is unlikely that the microbial population changed during the short incubation times required for measurable results by radioisotope assays (Kelly and Chynoweth 1980). Although the mercury concentrations used in our studies are greater than natural levels at SIL (Bodaly et al. 1984a), the amounts were not large enough to affect the microbial activity of the sediment (Furutani and Rudd 1980; Rudd et al. 1983; Ramlal et al. 1986). Therefore, the specific rates of methylation and demethylation are realistic; but, to convert them to absolute rates, the *in situ* concentrations of substrates, i.e. Hg^{2+} and CH_3Hg^+ would have to be known. Consequently, we use the M/D ratio to illustrate the relative balance of these rates among different sites and over time. Although M/D ratio were always less than 1.00 (e.g. Tables 1.5B and 2.4), this does not mean that absolute *in situ* demethylation rates exceed *in situ* methylation rates. The availability of substrate will also determine whether methyl mercury is accumulating or decreasing. The M/D ratio does show whether the organisms are capable of favouring either the methylation or demethylation reaction.

The mercury methylation and demethylation studies at SIL demonstrate the succession of methylating microflora which occurs after an area is flooded. Microorganisms in terrestrial environments often have limited access to water, nutrients and mercury, among other substances, which can be metabolized. In the upper organic soil layers of the shoreline around SIL, total mercury levels were much higher than in inorganic subsoil layers (Table 3.3; Bodaly et al. 1984a). If this mercury were methylated by soil microorganisms, the methyl mercury might either be released to the atmosphere (Friberg and Vostal 1972; Johnson and Braman 1974) or have a limited area of dispersal. No methyl mercury was detected in the soils used in the erosion experiments (Table 3.3) while small amounts were measured in moss-peat material used in the moss basket experiment (Table 3.2).

After an area of land is flooded, the terrestrial microhabitats of bacteria are either disrupted or completely destroyed, which will allow bacteria freer access to a wider variety of substrates. In soil, especially dry soil, microorganisms can persist in 'resting stages' until the habitat is more conducive to their growth (Atlas and Bartha 1981). The presence of water will often stimulate these resting stages, increasing the numbers of viable microorganisms (Atlas and Bartha 1981). An aquatic environment also makes substrates (e.g. nutrients) more accessible through hydration and diffusion. As the flooded terrestrial flora (trees, moss, etc.) undergo decomposition the amount of available bacterial substrate also increases. When these substrates are flooded, the microbial decomposition of this material can be quite rapid, releasing further nutrients.

In the comparison of SIL bank material and Wabigoon Lake sediment (Table 3.4) it was shown that the SIL material bound less mercury, and the mercury that was bound by SIL material was more easily released. Therefore, the addition of clay-material from the SIL banks to the lake would not have the same ameliorating effect as was found with the Wabigoon Lake sediment (Rudd and Turner 1983). They found that addition of Wabigoon Lake sediment to enclosures (limnocorrals) made ^{203}Hg less available for mercury methylation and bioaccumulation. In contrast Hecky et al. (1987) found that the addition of SIL bank material, increased mercury uptake by fish. This would mean that as long as the erosion of the banks continues, the bacteria will have available organic material, therefore, methyl mercury production can continue.

The results of the short term laboratory experiments using *Sphagnum* moss and ^{203}mercury (Table 3.1) indicate that moss can effectively reduce mercury methylation, while in this experiment mineral clays and lake water have a similar but much lower capacity to reduce methylation. The reduction in methylation presumably indicated that added ^{203}Hg is made unavailable to the microbes because of complexing or binding reactions with the added materials. However, in the moss basket experiment (Table 3.2) there was no detectable increase in either the total mercury or methyl mercury concentration over the duration of the experiment. This lack of evidence for significant binding of mercury is not surprising since the concentration of total mercury measured in the water of the SIL and Notigi reservoirs were less than the limit of detection (i.e. <5 ng $Hg \cdot L^{-1}$ in Bodaly et al. 1984a) which is less than that added in the laboratory experiments. Apparently, in the short term the moss can inhibit methylation possibly by sequestering dissolved mercury if it is at high concentrations, and it may not be a source of mercury. The inhibition of methylation observed experimentally (Table 3.1) may be of short duration because, as the moss is decomposed and used as a nutrient source by the microbiota, bound mercury can be released and may be methylated. The moss basket experiment would not detect such an effect because weight loss of the moss material over time was unknown.

Methyl mercury can be produced as an incorrect synthesis of the amino acid methionine (Landner 1971); and therefore, the amount of methyl mercury produced should increase as the microbial activity increases (Jernelov 1972). Pan-Hou et al. (1980) have noted that increased sensitivity to mercury by a vitamin B_{12} auxotroph (Clostidium cochlearium T-2C) was related to the loss of ability to methylate mercury. It is unlikely that there is a feedback mechanism controlling the production of methyl mercury since methyl mercury is not utilized by bacteria. Rather methyl mercury concentration will likely be controlled by the balance of methylating and demethylating bacteria and their respective microbial activity.

The limnocorral experiments showed how bacterial populations can be altered over longer time periods (months to years) by addition of mercury compared to that found in short term laboratory studies (days). All limnocorrals received a radioisotope spike of rather low specific activity. Consequently, they all experienced a pulse of mercuric ion (Hecky et al. 1987). Methylating activity was low in new controls; but when limnocorrals were used for more than one year in the same site (Table 1.4) an actively methylating population of bacteria was already established. In all the limnocorrals used more than once, the initial methylation measurements the following year were higher than in comparable new limnocorrals. In 1983, the mean M/D ratio for the season for all reused limnocorrals was at least three times higher than in new limnocorrals regardless of the treatments they were given. Even in the reused control limnocorral the methylation rate was high and the fish mercury levels were the highest recorded for the three years of the limnocorral experiment (Fig. 1.4). After two mercury additions, this limnocorral had high available mercury concentrations and a well established population of bacteria capable of producing methyl mercury.

The short term laboratory studies and longer term limnocorrals showed how the populations of mercury methylators can become established, sometimes quite rapidly. The speed of their establishment was treatment dependent, i.e. moss-peat limnocorrals had actively methylating populations within two months while the controls required two mercury spikes and at least one year (Hecky et al. 1987). The 1983 studies in Methyl Bay may be indicative of how the lake sediment bacterial population had reacted to several years of flooding. In the offshore, clay-rich sites which had always been under water, the M/D ratio was as low as in Granville Lake which was not flooded (Fig. 2.1B). However, in the flooded sites the M/D was considerably higher (Table 2.4). The bacterial flora was conditioned to produce methyl mercury at the flooded sites seven years after flooding. The microhabitat erosion experiments show that while both the clay sediment and the clay soil contained the same amount of total mercury, it was only the clay sediment, with an actively methylating population of bacteria, which had detectable methyl mercury in the sediment (Table 3.4). If the clay soil was inundated for a longer period, the bacterial population might methylate as fast or faster than the clay sediment.

The results of the first three years of mercury methylation studies show how the net methylation of mercury at SIL might have evolved after impoundment. The older flooded sites sampled (e.g. Methyl Bay) appear to have well established microbial populations which are capable of both methylation and demethylation. Over flooded terrain the balance of these two processes had shifted in favour of net methylation compared to offshore sediments present before impoundment. The two-year limnocorrals studies show how a population of methylating bacteria can be rapidly selected if treatments supply substrate which favours rapid growth of bacteria, e.g. new moss treatments in 1981 and 1982, or slowly if only Hg^{2+} is added, e.g. in reused controls. With rapid growth the selection for methylating microorganisms can occur quickly because of the greater number of generations produced in a short time period, but the population will probably fluctuate until the conditions of the habitat are at a more constant state. Demethylation of methyl mercury seems to be regulated by factors other than the nutrient supply, since it did not follow any definable trend. The process is known to be plasmid mediated (Pan-Hou et al. 1980) and the mercuric reductase-plasmid bearing populations may not be fully established in the limnocorrals, or, the populations may be controlled by other factors in addition to methyl mercury concentrations.

Because the limnocorrals were not microbiologically in a constant state, the methylation rates measured did not always reflect the total mercury levels in the fish of these enclosures (Figs. 1.2-1.4). However, in the mercury survey of 1982 the concentration of mercury in fish muscle followed the same trends as the rate of sediment mercury methylation at the sites which had been flooded for six years (Fig. 2.1).

This research does illustrate that it is necessary to have paired methylation and demethylation to determine how the microbial population is responding to changes. Low M/D ratios would indicate that the probability of net mercury methylation was low (unless Hg^{2+} concentrations were very high) and methyl mercury would probably not be a problem. This study also shows that the amount of organic material available to the bacteria will regulate the amount of methyl mercury produced. Present research is directed towards comparison of unflooded and flooded sites in a number of lakes, similar to measurements made in Methyl Bay, so that comparison of pre-and post-impoundment sediments can be made.

ACKNOWLEDGMENTS

We thank the many people at the SIL fisheries camp for their help and cooperation in collecting field samples. F.A.J. Armstrong, R.A. Bodaly, G.J. Brunskill, P. Campbell, M. Capel, R.H. Hesslein, S.G. Lawrence, and K. Marshall, considerably improved the manuscript with their suggestions. Total mercury analyses were done by R. Hunt and sediment chemistry was done by B. Hauser. We also thank the Canadian Department of Fisheries and Oceans, and the

Canada-Manitoba Mercury Agreement for funding this research. Aspects of this study were conducted by Agassiz North Associates under DSS contract 0SF83-0004.

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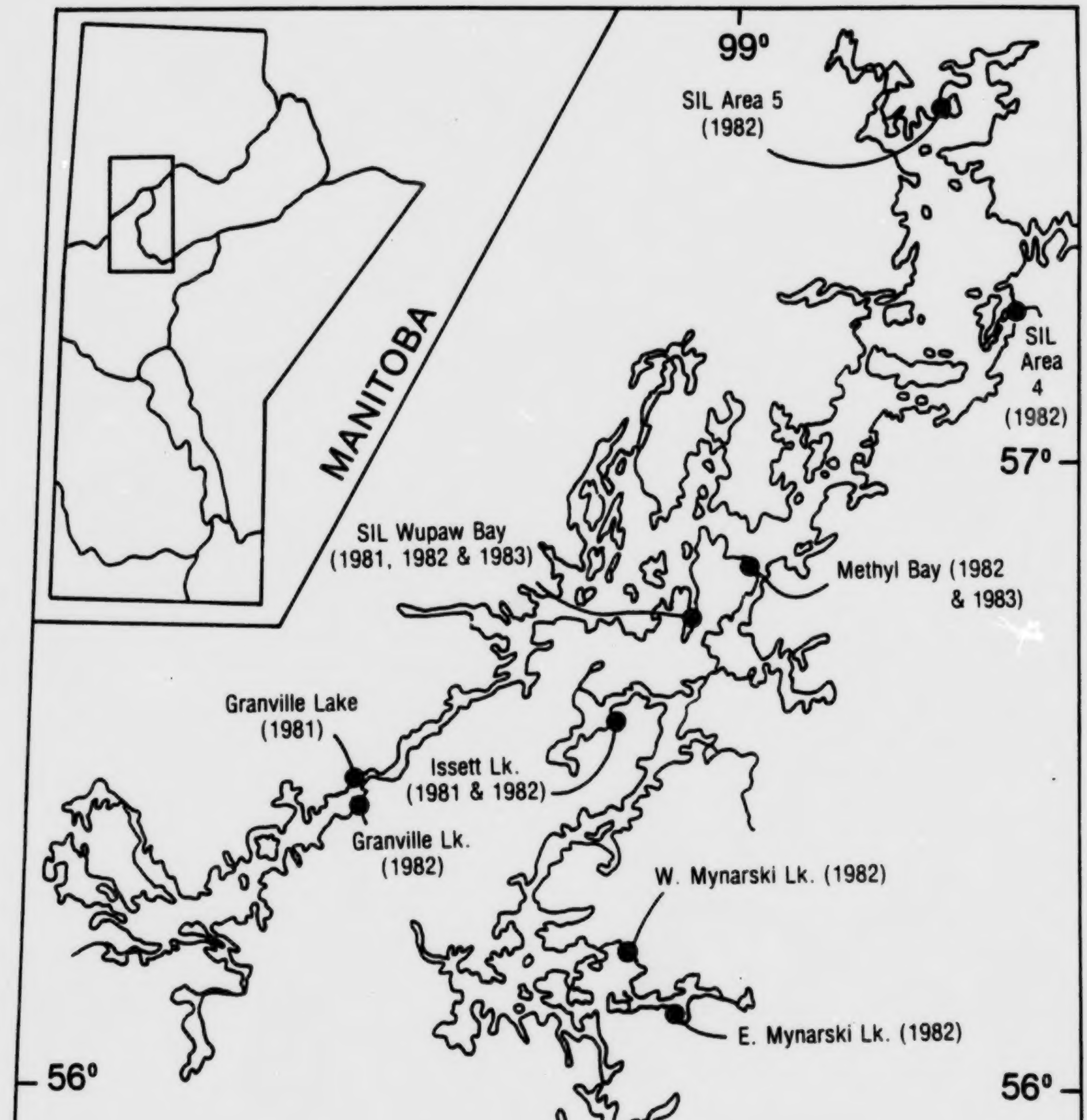
MANITOBA
99°
SIL Area 5
(1982)
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SIL Wupaw Bay
(1981, 1982 & 1983)
Methyl Bay (1982
& 1983)
Granville Lake
(1981)
Issett Lk.
(1981 & 1982)
Granville Lk.
(1982)
W. Mynarski Lk. (1982)
E. Mynarski Lk. (1982)
56°
56°
Notigi Lk. West
(1982)
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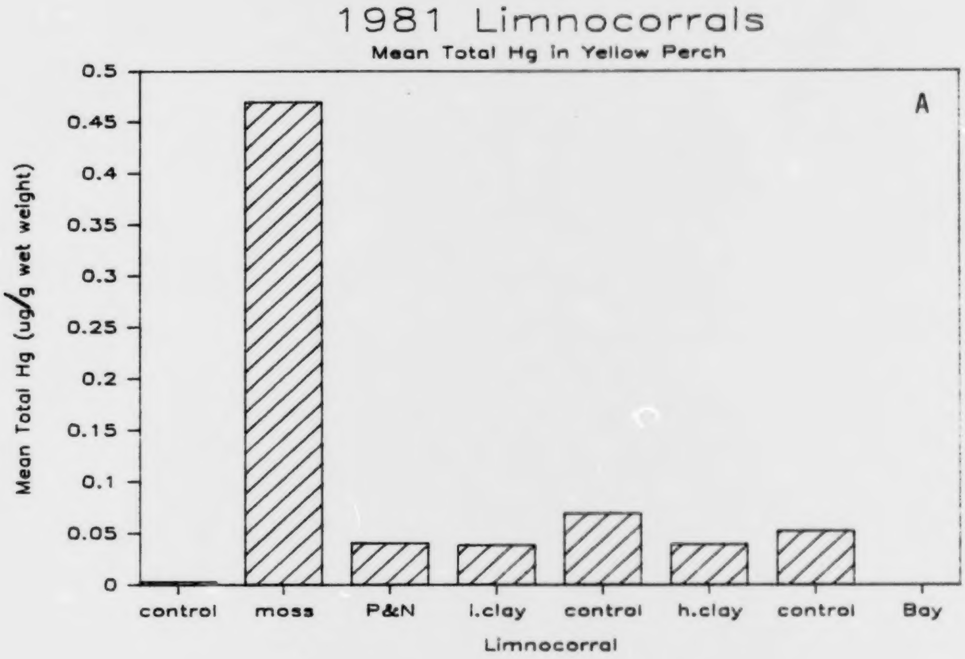
Fig. 1.1. Map of Southern Indian Lake and Notigi reservoirs.



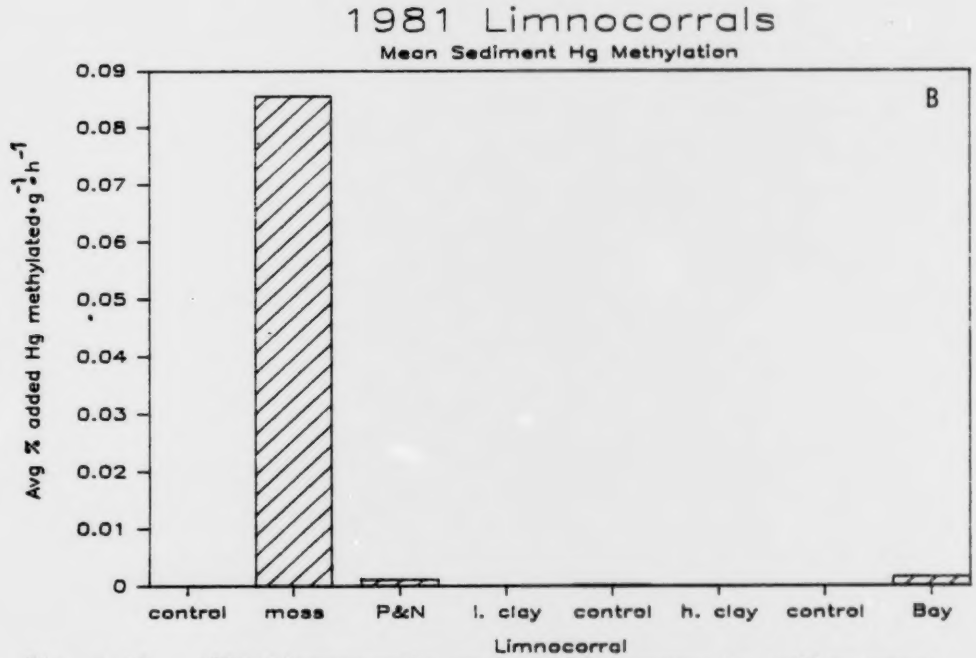
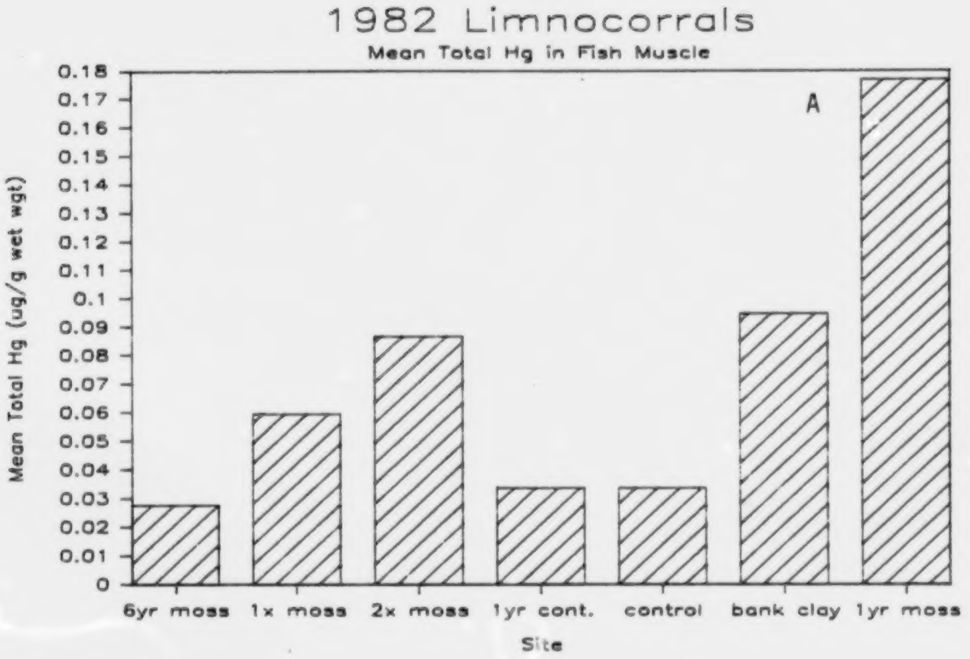


Fig. 1.2A. 1981 limnocorrals-mean total mercury in yellow perch, August 1981 (Hecky et al. 1987).

Fig. 1.2B. 1981 limnocorrals-mean sediment mercury methylation rates. (There are no data available for the samples registering zero on the y-axis.).



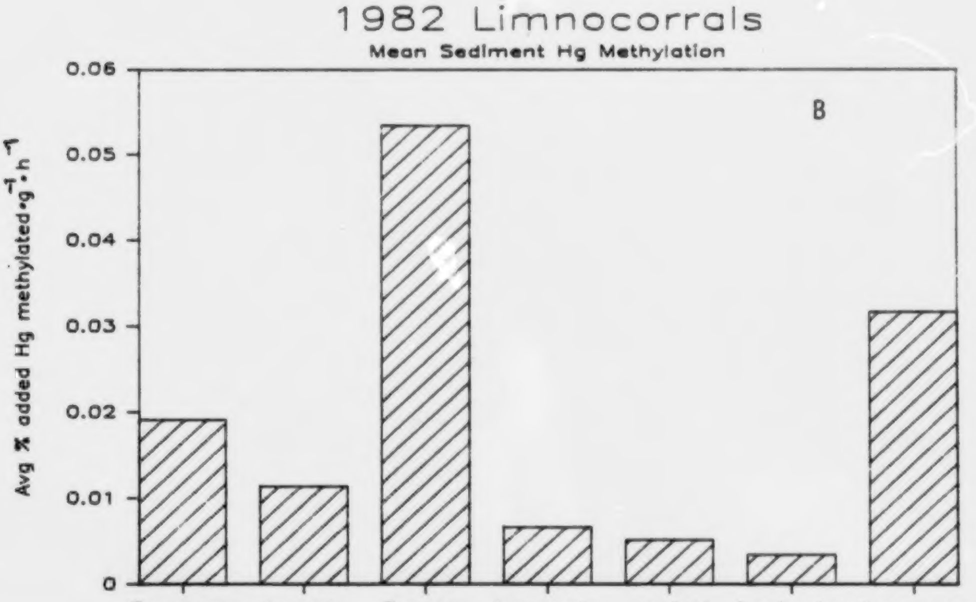
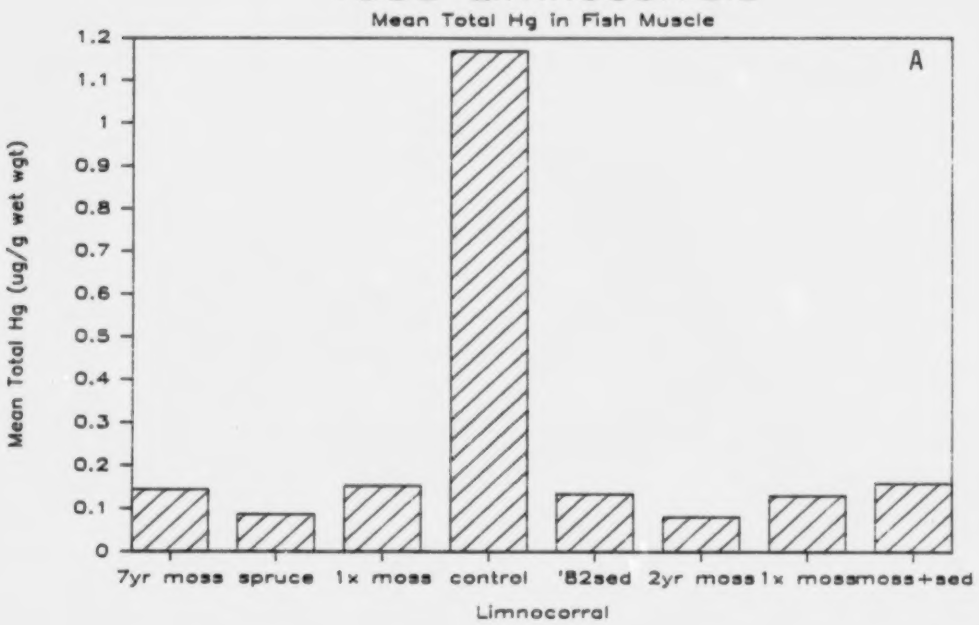


Fig. 1.3A. 1982 limnocorrals-mean total mercury in fish muscle at the end of the experiment (Hecky et al. 1987). There are no data available for the samples registering zero on the y-axis.

Fig. 1.3B. 1982 limnocorrals-mean sediment mercury methylation rates.



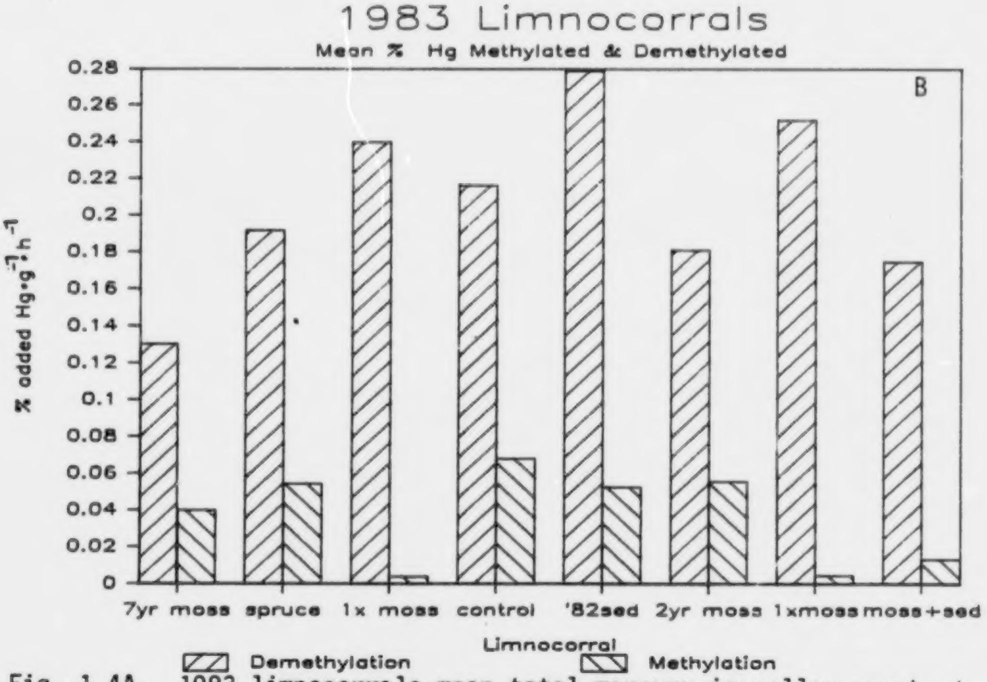
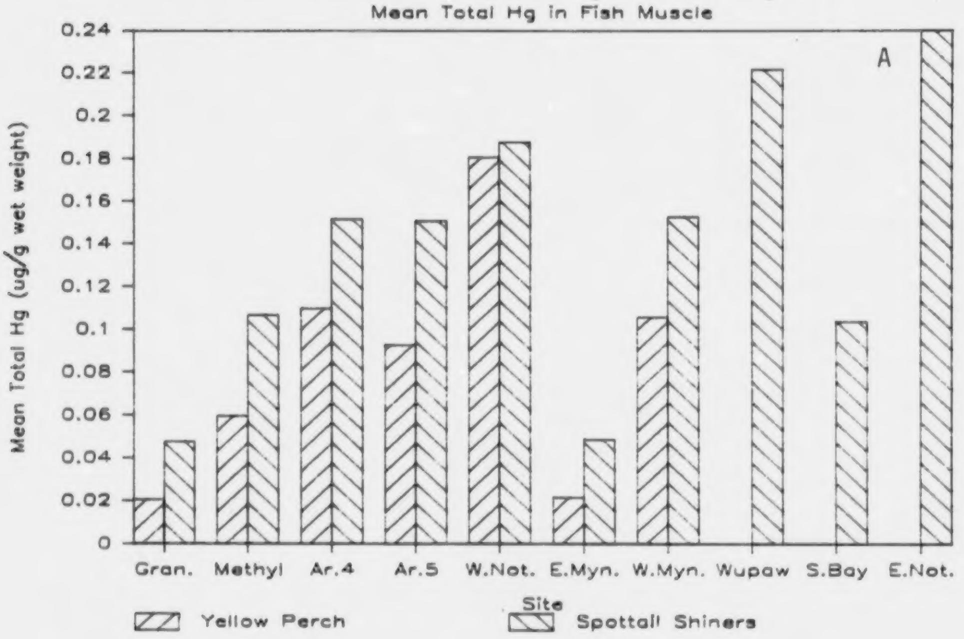


Fig. 1.4A. 1983 limnocorrals-mean total mercury in yellow perch at the end of the experiment (Hecky et al. 1987).

Fig. 1.4B. 1983 limnocorrals-mean sediment mercury methylation and demethylation rates.



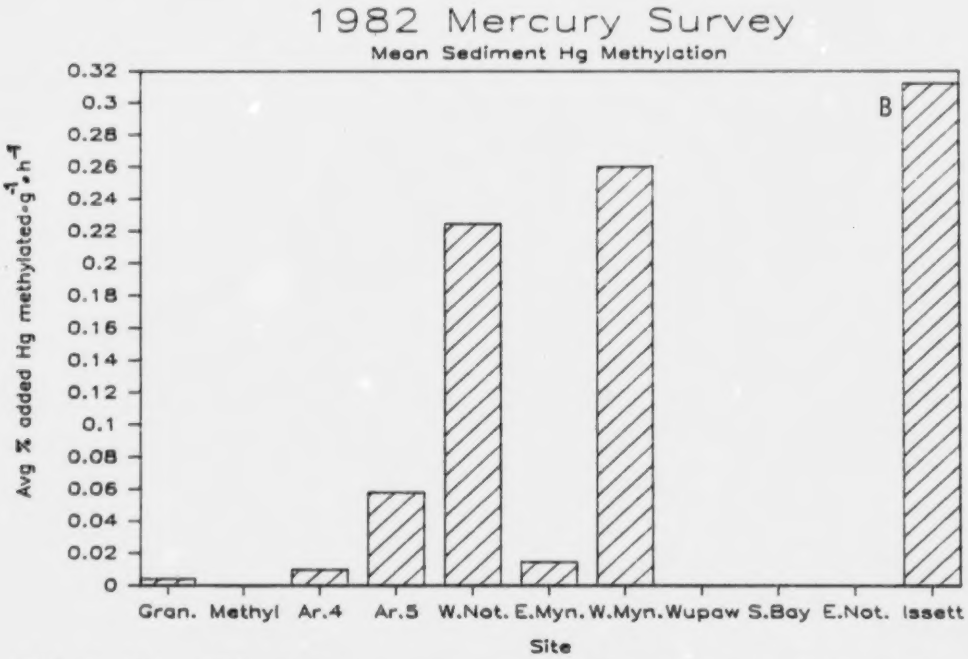


Fig. 2.1A. 1982 mercury survey-mean total mercury in fish muscle (Bodaly et al. 1987).

Fig. 2.1B. 1982 mercury sruvey-sediment mercury methylat rates. West Notigi is given at one-tenth its actual value.

APPENDIX: SEDIMENT CHEMISTRY

Table A1. Chemistry of sediment used for limnocorral methylation studies, 1982. C-carbon and N-nitrogen are expressed as mg$\cdot$g^{-1} dry weight; R-respiration as µM $O_2 m^{-3} \cdot h^{-1}$.

Limnocorral	July 8			July 21			August 3			August 23		
	C	N	R	C	N	R	C	N	R	C	N	R
6 yr moss	26	2.8	787	47	4.8	1097	136	9.5	1838	93	7.5	914
1x moss	70	10.1	826	43		986	96	9.1	1456	85	6.7	686
2x moss	19	2.2	1079	53		2058	145	8.3	2669	96	6.1	1644
1 yr cont.	48	7.4				1222	105	8.3	447	63	5.3	810
new cont.	51	4.9	1303	35	5.5	670	119	8.6	875	84	7.9	
bank clay	42	5.5	791			821	57	5.6	385	44	3.8	
1 year moss	51	5.4	2462			1129	99	6.1	490	60	6.0	

Limnocorral	Mean Values		
	C	N	R
6 yr moss	76	6.2	1159
1x moss	74	8.6	989
2x moss	78	5.5	1863
1 yr control	72	7.0	826
new control	72	6.7	949
bank clay	48	5.0	666
1 yr moss	70	5.8	1360

Table A2. pH and rates of carbon dioxide production per gram dry weight of sediments collected from limnocorrals at SIL during 1983.

Limnocorral	July 10, 1983 $\mu mol CO_2 \cdot g^{-1} \cdot h^{-1}$	pH	July 25, 1983 $\mu mol CO_2 \cdot g^{-1} \cdot h^{-1}$	pH	August 7, 1983 $\mu mol CO_2 \cdot g^{-1} \cdot h^{-1}$	pH
7 yr moss (disturbed)	3.08	6.42	0.35	6.60	<0.10	6.66
spruce	0.25	6.54	0.91	6.08	1.10	6.31
1 x moss	0.74	6.97	0.34	6.81	<0.10	6.88
control	0.32	6.69	0.38	6.81	<0.10	6.70
1982 clay addition	-	7.59	0.43	6.72	<0.10	6.70
2 yr moss	<0.10	7.26	0.81	6.68	<0.10	6.69
1x moss	2.53	7.70	<0.10	6.82	<0.10	6.79
1x moss + lake clay	6.84	7.45	0.28	6.83	<0.10	6.85

Table A3: Chemistry of limnocorral sediment expressed as mg carbon (C) or nitrogen (N) per gram dry weight used in mercury methylation studies, 1983.

	July 10		July 25		August 7		Average	
Limnocorral	C	N	C	N	C	N	C	N
7 yr moss (disturbed)	160	9.7	133	8.2	167	10.5	153	9.5
spruce	985	7.6	132	9.8	102	8.5	406	8.6
1 x moss	49	3.0	47	3.0	52	3.5	49	3.2
control	109	12.4	100	8.8	122	9.3	110	10.2
1982 clay addition	74	4.7	72	5.6	72	5.9	73	5.4
2 yr moss	116	7.5	105	7.2	85	7.1	102	7.3
1x moss	49	3.5	53	4.1	63	4.2	54	3.9
1x moss + lake clay	64	4.5	68	4.6	51	3.3	61	4.1

Table A4: Chemistry of sediment used for methylation studies in Methyl Bay, SIL 1982. C-carbon and N-nitrogen expressed as mg$\cdot$g dry sediment^{-1}; R-respiration as $\mu M\ O_2 \cdot m^{-3} \cdot h^{-1}$.

	July 6				July 19	August 16		
Site	C	N	R	R	R	C	N	R
clay			1170	1027	1405	25	2.7	1433
flooded	57.1	7.5	775	1414	1478	54	5.1	1598
flooded			475	517	1883	90	4.2	1498
clay	44.6	6.3	773	512				
clay				1233	3159	14	1.7	3635

	Average Values		
Site	C	N	R
clay	25	2.7	1259
flooded	56	6.3	1316
flooded	90	4.2	1093
clay	45	6.3	643
clay	14	1.7	2676

Table A5: Chemistry of mercury survey methylation samples, August 17, 1982.

Site	C ($mg \cdot g^{-1}$)	N ($mg \cdot g^{-1}$)	Respiration ($\mu M\ O_2 \cdot m^{-3} \cdot h^{-1}$)
Mynarski E.	80	6.4	1422
Mynarski W.	197	18.1	3084
Notigi W.	73	5.8	2389
Area 5	114	11.8	1429
Area 4	75	3.5	1227
Granville	12	1.8	7855
Isset	98	9.4	

Table A6: Chemistry and rate of carbon dioxide production of sediments (expressed µmol·C·g dry weight^{-1}·h^{-1}) collected from Methyl Bay, SIL during 1983. C-carbon and N-nitrogen expressed as mg·g·dry sediment^{-1}.

	July 19, 1983				August 1, 1983			
Station	CO$_2$ Prod'n	pH	C	N	C[illegible] Prod'n	pH	C	N
clay	0.58	6.5	32	3.1	<0.10	6.8	29	2.3
flooded	2.78	6.7	135	5.8	1.39	6.5	85	5.9
flooded	0.11	6.9	99	5.5	<0.10	6.7	178	6.7

Volume

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Canada Manitoba

6. Enhancement of Mecury Bioaccumulation in Fish by Flooded Terrestrial Materials in Experimental Ecosystems
   R.E. Hecky, et al

7. Mercury Bioaccumulation in Yellow Perch in Limnocorrals Simulating the Effects of Reservoir Formation
   R.E. Hecky, et al

8. Methylation, Demethylation, and Bioaccumulation of Mercury in Lakes and Reservoirs of Northern Manitoba, with Particular Reference to Effects of Environmental Changes caused by the Churchill Nelson River Diversion
   T.A. Jackson

Canada - Manitoba Agreement on the Study and Monitoring of Mercury in the Churchill River Diversion

Technical Appendices to the Summary Report Volume 2

ENHANCEMENT OF MERCURY BIOACCUMULATION IN FISH BY FLOODED TERRESTRIAL MATERIALS IN EXPERIMENTAL ECOSYSTEMS

A Summary Report for the Canada-Manitoba Agreement on the Study and Monitoring of Mercury in the Churchill River Diversion:
Project 2.1 Experimental Ecosystems

by

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SPECIAL NOTE

This report was prepared for the Canada-Manitoba Agreement on the Study and Monitoring of Mercury in the Churchill River Diversion. Any conclusions, recommendations, or opinions expressed herein are those of the authors and are not necessarily the same as those of the Steering Committee for this Agreement.

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EXECUTIVE SUMMARY

1. A series of limnocorral experiments to investigate the effect of reservoir formation on fish mercury concentrations were conducted from 1981-1984 at Southern Indian Lake. These experiments were funded by the Canada-Manitoba Agreement on the Study and Monitoring of Mercury in the Churchill River Diversion and the Department of Fisheries and Oceans. The experiments were specifically formulated to investigate whether the terrestrial materials added to northern reservoirs and lakes were stimulatory to bioaccumulation of mercury in fish.

2. The organic materials investigated were moss-peat, the predominant soil and ground cover in the low lying areas of northern Manitoba, and black spruce boughs, the dominant tree. Inorganic materials tested were the dissolved inorganic nutrients, nitrogen and phosphorus, eroded bank clay, and lake sediment. Prairie sod was also tested to determine if the effects of flooding were specific to the boreal forest. The organic materials and sod were added at one time only to limnocorrals. The amounts of material added were chosen to match the flooded material:water volume ratio of Notigi Reservoir. The bank clay and lake sediment were added repeatedly in suspension at concentrations similar to those observed in Southern Indian Lake after impoundment. The inorganic nutrients increased and maintained the primary productivity of the treated limnocorral at levels similar to those observed in Notigi Reservoir after impoundment. In addition, the effect of different concentrations of mercuric chloride added as a spike were investigated.

3. The effect of the treatments on the mercury content of yellow perch was followed over a 60-90 d period by sampling fish at one to three week intervals depending on the year. The mercury accumulating in the muscle of yellow perch

was found to be over 90% methyl mercury. The amounts of mercury accumulated by yellow perch were similar to the amounts accumulated by other small fish, e.g. emerald shiners in these limnocorrals.

4. The addition of the radioactive ^{203}Hg isotope as mercuric chloride at the beginning of the experiment allowed estimation of the amount of inorganic mercury made available with the treatments in several experiments. Mercuric ion available for methylation in these experiments was always small, <1% of the total mercury added with treatments; but the amount of mercuric ion from added materials and the natural substratum of the limnocorrals was always large relative to the mercuric ion added with the radioisotope spike.

5. Over the two to three month course of the experiments, there was no relationship between the final concentration of mercury in yellow perch muscle and the following diverse measures of total mercury in the limnocorrals: 1) total mercury added with a treatment, 2) total mercury added as mercuric chloride as a spike, 3) the concentration of mercury in the treatment material, and 4) mercuric ion "available" for methylation. Likewise, using the ^{203}Hg isotope to follow the ramification of added mercury into the various chemical and biotic forms of mercury in the limnocorrals, there was no correlation between these different forms and the mercury concentration in yellow perch muscle.

6. Moss-peat material was added to limnocorrals every year for four years; and in every year it increased mercury concentrations in yellow perch muscle relative to contols. At a loading rate similar to that which occurred in Notigi Reservoir, the moss-peat material doubled the mercury concentration of the controls in every year except 1981 when it increased the fish mercury by a

factor of four. Doubling this loading rate in 1982 approximately doubled the amount of mercury accumulating in fish muscle.

7. Black spruce boughs increased fish mercury to an even greater extent than moss-peat. Therefore, the most common organic materials in the boreal forest stimulated the bioaccumulation of mercury by yellow perch at rates of addition of these materials which occur in new reservoirs. At least in the case of moss-peat material the effect is additive with a doubling of the amount added doubling the amount of mercury accumulating in fish.

8. Prairie sod had the same effect on body burden of mercury as did the moss-peat and spruce indicating that it stimulated methyl mercury production to the same extent as the other materials. However, the prairie sod also greatly stimulated fish growth and the net effect was to produce lower concentrations of mercury in fish muscle relative to the boreal forest materials, but these concentrations were still well above the fish mercury concentrations in the control limnocorrals.

9. Added suspended sediments, whether from eroding bank clays or lake sediments, were either neutral or stimulatory to mercury accumulation in fish. Eroded bank clays, when added in suspension to a limnocorral with natural lake sediments, were as stimulatory as moss-peat by the end of the experiment although the increase occurred later. When the same bank clay was added to plastic-bottomed limnocorrals, the positive effect was not apparent suggesting that the microbial populations and bacterial substrate in the natural sediments were essential for the positive effect. The bank clay added in suspension could have increased the supply of mercuric ion and/or surface area for bacterial colonization thereby stimulating methyl mercury production and accumulation in fish.

10. Stimulation of microbial activity and consequently methylation of mercury by flooded terrestrial materials is the cause of the increased mercury concentrations in fish in limnocorrals in nature. Consequently, the return of mercury concentrations in fish to their pre-impoundment levels in flooded lakes will be dependent on the time course for degradation of the added materials. Some of the flooded organics, e.g. tree boles, breakdown very slowly; therefore, microbial activity in reservoirs may remain elevated for a long time. As most of the added terrestrial materials occur in shallow waters where sedimentation rates are generally low, burial of the flooded material beneath a layer of sediment will not hasten this process except in unusual circumstances where there is a local source of sediment near a protected site. In this case, wave action will not remove the sediments as they are deposited. The flooded land surface in the protected bay where the experiments were conducted may have been an example of such a situation.

11. The addition of suspended sediments, e.g. by dredging, has been recommended as a mitigation procedure in other mercury contaminated systems. However, the addition of suspended sediments to a moss-peat treated limnocorral did not reduce mercury uptake by fish within the two month period of the experiments. Thus, the addition of suspended sediments to the Churchill River Diversion reservoirs cannot be recommended.

INTRODUCTION

The Churchill River diversion caused extensive flooding on Southern Indian Lake and along the diversion route in the Rat and Burntwood River valleys in northern Manitoba (Newbury et al. 1984). Subsequent to lake impoundment and river diversion, mercury concentrations in the commercial catch of northern pike and walleye from Southern Indian Lake rose above the acceptable limit for marketing fish in Canada and approached the United States marketing limit. In other lakes flooded by the Churchill River diversion even higher mercury concentrations occurred (Bodaly and Hecky 1979; Bodaly et al. 1984a). Because nearly all the mercury in fish muscle is in the form of methyl mercury (Westoo 1966; Lockhart et al. 1972) which is a powerful neurotoxin, elevated fish mercury concentrations can be a human health hazard if sufficient quantities of fish are consumed. Recognition of the mercury problem in northern Manitoba resulted in a claim being filed under the Northern Flood Agreement and the signing of the Canada-Manitoba Agreement on the Study and Monitoring of Mercury in the Churchill River Diversion. This agreement identified the need for numerous projects including the one reported on here, 2.1 Experimental Ecosystems.

As stated in the Canada-Manitoba agreement, the objective of this project is "....to study the effects of changes associated with lake impoundment on the bioaccumulation of mercury....The environment will be changed within the corrals to simulate changes which occur in lakes when they are flooded. The mercury content of fish within the limnocorrals will then be related to the changes in the enclosed environment." The research of this project is also directly relevant to the Arbitrator's interim consent order to Claim 12 under the Northern Flood Agreement calling for studies "....to ascertain the source of any increased levels of mercury in fish....in the project area, to ascertain the mechanisms by which mercury enters the food chain in the aquatic environment of the Project, and to evaluate the likely effectiveness of any potential remedial or mitigatory measures to reduce the entry of mercury into the food chain in that environment." The experiments conducted specifically examined the effect on mercury bioaccumulation in fish of various terrestrial materials transferred to the aquatic environment by impoundment, the mercury yield from those materials into fish, and one possible mitigation action.

THE MERCURY CYCLE IN LAKES AND RESERVOIRS

Mercury accumulates in fish tissue predominantly in the form of methyl mercuric ion hereafter referred to as methyl mercury. This compound is an extreme neurotoxin, and its concentration in fish muscle is a health concern for human consumption of fish. Microbial methylation is the overwhelming source of methyl mercury production; autoclaved water or sediment have essentially no methylation activity relative to non-autoclaved samples (Furutani and Rudd 1980; Xun et al., in press). Bacteria can only methylate free mercuric ion, and the methylation reaction is thought to be either a detoxifying reaction (Pan-Hou and Imura 1982) or an incorrect synthesis of organic acids such as amino acids (Tonomura and Kanzaki 1969). Some bacteria can also demethylate methyl mercury (Tonomura and Kanzaki 1969; Wood et al. 1972). Consequently methyl mercury concentrations in nature are controlled to a large extent by mercuric ion availability and bacterial activity (Fig. 1).

Methyl mercury is taken up more rapidly and released more slowly than inorganic mercury by aquatic organisms (Huckabee et al. 1979). Aquatic invertebrates absorb and eliminate methyl mercury faster than fish; consequently the methyl mercury to total mercury ratio in invertebrates is much lower than in fish, especially fish muscle (Huckabee et al. 1979). Fish take up methyl mercury both from the water through their gills and from their food as well as methylating mercury in their guts (Rudd et al. 1980). Among other factors, fish age and piscivory are almost always positively correlated with muscle mercury concentrations within any aquatic system (Scott and

Armstrong 1972; Scott 1974; Huckabee et al. 1979). Fish muscle tissue and methyl mercury have a high affinity for each other and water to fish muscle concentration factors in excess of 600,000 have been demonstrated in Southern Indian Lake as well as other field and laboratory studies (Bodaly et al. 1984a; Ribeyre and Boudou 1984). Because of these high concentration factors, total and methyl mercury concentrations in water can be below analytical detection limits while mercury in fish muscle exceeds the accepted marketing limits.

Reservoirs can affect the aquatic mercury cycle in several different ways. The most obvious is by introducing mercury incorporated in terrestrial materials into the aquatic system thereby increasing the loading of mercury to the reservoir water. Terrestrial organic materials generally have total mercury concentrations as high or higher than natural lake sediments in the Churchill River diversion area (Bodaly et al. 1987). Consequently, the mean concentration of total mercury on the lake bottom will be higher after impoundment. However, the extent of release of mercury from such inundated materials has to be determined, i.e. whether mercury from flooded materials significantly increases the amount of methylatable mercuric ion present in the system. Even if all the Hg in inundated terrestrial material were insoluble, the effect of the added material on water quality might alter the equilibria controlling mercuric ion. Benes and Havlik (1979) have reviewed the speciation of Hg in natural waters. They concluded that knowledge of speciation even in simple systems is "still very imperfect", and this is even more so in reservoirs where complex water quality changes can occur, e.g. higher or lower suspended sediment, dissolved oxygen, dissolved organic matter concentrations, pH, etc. In addition, reservoirs create quite different, non-steady state, water quality conditions in the vicinity of flooded soils and vegetation compared to the conditions in the open water areas. Therefore, there is no theoretical model to predict the overall effects of reservoir formation on mercuric ion or methyl mercury availability. Given the number of mercury compounds present in natural waters and the complexity of possible transformations, this situation is not likely to change in the near future; empirical approaches and solutions will have to suffice. The greatly simplified representation of the inorganic mercury cycle in Fig. 1 serves only to illustrate the predominate inorganic forms to be found in solution. In fact, suspended particulate species are the most abundant forms of mercury in all surface waters.

Reservoir formation generally causes a trophic upsurge, i.e. a period of increased biological activity and growth in all levels of the food chain. However, the specific characteristics of a reservoir can greatly modify the upsurge. In the Churchill River diversion, Southern Indian Lake did not exhibit such an upsurge (Hecky et al. 1984) while the Notigi Reservoir followed a more classical reservoir response (Bodaly et al. 1984b). Increased biological activity in new reservoirs should affect the biological cycling of mercury (Fig. 1). Most importantly the stimulation of bacterial activity by the availability of flooded terrestrial organics could alter the methyl mercury concentrations throughout the whole reservoir or perhaps just in the vicinity of flooded vegetation. As bacteria can both methylate and demethylate mercury and the conditions favoring one or the other are imprecisely known (Bisogni 1979), it is difficult to predict *a priori* how reservoir formation will affect net methylation of mercury. Also, effects on net methylation may not be required to induce changes in fish muscle mercury concentrations. For example, if net methylation were unchanged but fish grew faster in new reservoirs as is frequently observed during the trophic upsurge, mercury concentrations could even decline as growth increased faster than mercury uptake from water and food. The converse, lower fish growth with unchanged methyl mercury concentrations could lead to an increase in mercury concentrations in fish muscle. In addition, water quality has been shown to affect the direct uptake of methyl mercury over the gills (e.g. temperature - Reinhard et al. 1974; water hardness - Rodgers and Beamish 1981) so that changing water quality might affect bioaccumulation in fish without changing methyl mercury concentrations in water.

EXPERIMENTAL APPROACH

The complexity of the biogeochemical cycle of mercury, the diverse effects that reservoir formation can have on that cycle, and the difficulty of measuring the concentrations of the relevant dissolved mercury compounds in the lakes and reservoirs of the Churchill River diversion all pose obstacles to understanding why mercury concentrations in fish flesh have increased. Identification of significant causal factors required an empirical approach with simplification and control of as many aspects as possible. Rudd et al. (1983) had demonstrated the utility of experimental limnocorrals (large enclosures to isolate a defined water mass from a larger body of water) in understanding mercury cycling in the industrially polluted English-Wabigoon River system. It was decided to follow this approach in regard to the mercury problem in the Churchill River diversion. Advantages of this approach are: 1) control of several significant factors, 2) specificity of treatment and response, 3) appropriate scaling, 4) addition of a radioisotope allows transformations to be followed and permits estimation of natural mercury availability, and 5) most importantly, applicability.

All the limnocorral experiments are controlled in the sense that nearly identical water volumes and sediment areas are isolated from a relatively homogeneous water body. Some are treated, and some are not (controls). All observations of treatment effects are made relative to controls. In studying impacted lakes or reservoirs the establishment of a suitable control (a similar presumably non-impacted system) is always problematical. Although laboratory experiments also offer control, they present scaling problems, e.g. high surface to volume ratios of containers can stimulate bacterial growth, difficulties in presenting all natural substrates, processes and microhabitats, difficulties in maintaining natural communities, etc., which often make it difficult to apply the results to the natural system of interest. In reservoirs, a diversity of terrestrial materials are being added concurrently, and it is impossible to separate the effects of these different materials on mercury cycling. In limnocorrals the effects of defined amounts of individual materials can be established. Finally, the availability of a radioisotope of mercury makes it possible to follow the transformation of the added mercury into different components of the ecosystem. If the radioisotope comes into equilibrium with the mercuric ion available for methylation, it is possible to estimate the amount of mercuric ion available and thereby the yield of mercuric ion from added materials.

This report is an interpretive summary of an original research report by Hecky et al. (1987). Four years of limnocorral experiments are reported here. The 1981 and 1984 experiments were solely funded by the Department of Fisheries and Oceans while the 1982 and 1983 experiments were jointly funded by the Canada-Manitoba Agreement and the Department. The reader should consult the original report for detailed explanations of methods and results. As that report will be published along with this summary, tables, figures and appendices from that report will not be reprinted here in order to save space and expense. Reference to tables, figures and appendices in that report will be made by prefacing the reference with a D in this report, e.g. Figure 1 of that report will be Figure D1 of this report, Table 1 would be Table D1 and Appendix Table A1.1 would be DA1.1

METHODS

Limnocorral experiments were conducted in the summers of 1981, 1982, 1983, and 1984. Except as noted here or in Hecky et al. (1987), effort was made to construct and manage the limncorrals in the same manner in every year so that experiments from different years could be compared. The limnocorrals were 10 m in diameter and were installed in water 2 to 4 m in depth (Table D1) in Wupaw Bay on Southern Indian Lake. Bodaly and Lesack (1984) describe the bay and its fish community and limnology. The limnocorral walls were translucent, fibre-reinforced polyethylene. The bottoms of the limnocorrals were either of the same material (1981 experiments only) or lake sediments (a silty clay) in which case the wall material was sealed to the sediment surface by placing sandbags on excess wall material. The limnocorrals within any year were installed over a period of two to three days in the latter half of June.

After installation, the limnocorrals were fished with a variety of gear including fine mesh seine and gill nets and minnow traps in an attempt to remove all fish caught by the installation. These efforts were always successful for larger predatory fish, but smaller fish often remained. Yellow perch (usually age 1+) were then introduced into the limnocorrals (emerald shiners were also added in 1981 only). These fish were captured by seining in the vicinity of the limnocorrals and usually placed into the limnocorrals immediatey after transport from the seining areas. Stocking density varied somewhat from year-to-year especially in 1983 when it was difficult to catch yearling yellow perch in the bay and two year old fish were stocked (Table D3). Growth of fish in limnocorrals was as good as (or better than) growth of wild fish in the bay over the same period. Survivorship was good in every year except 1981, but adequate numbers of fish for analysis were available in every year. A subsample of the fish added to the limnocorrals was taken for a time zero determination of length, weight and total mercury content (Appendix DA3).

The experimental treatments chosen were based on observed water quality changes in the Southern Indian Lake (Hecky et al. 1984) and Notigi Reservoirs (Bodaly et al. 1984b). The experimental treatments (Tables D1 and D2) were made after extraneous fish removal and before stocking with perch. The organic treatments were generally scaled to the flooded area to volume ratio of Notigi Reservoir (Newbury et al. 1984) using surveys of vegetation to determine biomass per unit area around Wupaw Bay (J.M. Shay, unpublished data) which is tyical for the boreal forest in northern Manitoba. For example, the amount of moss and peat material to be added to a limnocorral of known volume was calculated by estimating the amount of moss and peat flooded (assumed to be on average a 5 cm thick layer over the area flooded) in the Notigi Reservoir and dividing by the volume of the Notigi Reservoir. This loading rate would then be applied to the limnocorral. Suspended sediments were added in sufficient quantity to maintain an average Secchi disk reading similar to that occurring over much of Southern Indian Lake after flooding (Hecky 1984). The 1981 nutrient addition was sufficient to boost primary productivity to a level characteristic of Notigi Reservoir after impoundment (Bodaly et al. 1984b).

Radioactive ^{203}Hg as $HgCl_2$ and tritiated water were added as one-time spikes to the limnocorrals after experimental treatments were made and yellow perch had been stocked. Details of the isotope additions are found in Table D5. The tritiated water allowed an initial estimate of the volume of each limnocorral and was monitored throughout the experimental period to check for leakage from the limnocorrals. Because the ^{203}Hg added was not carrier free and in fact, had a rather low specific activity, a substantial amount of inorganic Hg was added with each spike. The total Hg concentrations in Wupaw Bay waters, as throughout most of the Churchill River diversion area, were below detection limits (Bodaly et al. 1984a; Williamson 1986) but were probably in the range of 1-2 $ng \cdot L^{-1}$. The radioisotope spike increased the total Hg concentration in the water immediately after the spike to 2-8 $ng \cdot L^{-1}$ depending on the year and the limnocorral volume (Table D5). In 1984, two limnocorrals were run without any radioisotope addition to determine if the spiking with Hg may have influenced results. The conclusion was that the mercury spike had no significant effect on mercury mobilization to fish in first year limnocorrals.

After the radioisotope spike, samples of fish were taken at regular intervals over the course of the experiment. Sampling intervals varied from triweekly in 1983 to weekly in 1984. At the end of the experimental period, near the end of August, except in 1984 when sampling was extended through September, all the fish in the limnocorral were recovered by treating the limnocorrals with rotenone. From these periodic samplings of fish, the uptake of radioactive Hg and total Hg could be followed to determine the effect of a treatment relative to controls. These data on the fish are reported *in extenso* in Appendix DA3. A number of other measurements of the concentration of the mercury isotope in different biogeochemical forms in the water were followed over time in one or more years (Appendix DA2). Radioisotope measurements were made on some of the other components of the limnocorral systems, e.g. wall material, sediments, benthic and planktonic organisms and these are reported in Appendix DA4. The physical conditions in the limnocorrals, i.e. temperature, conductivity and transparency, were monitored in all years (Appendix DA1) while extensive chemical data were available for 1981 to 1983 (Appendix DA5).

RESULTS AND DISCUSSION

Each treatment made to a limnocorral tested an hypothesis about the effect of that treatment on mercury accumulation in fish. Over four years, 21 different treatments were applied, but the hypotheses tested can be conveniently grouped into 1) the effects of added dissolved Hg, 2) the effect of terrestrial organic materials, 3) the effects of suspended sediments, and 4) the effect of dissolved inorganic nutrients. The data report of Hecky et al. (1987) discusses the experiments in chronological order, but the following discussion will group the experiments by the general hypotheses regardless of year.

ADDED DISSOLVED Hg

Aquatic systems contaminated by industrial mercury have been the subject of intensive studies (e.g. the English-Wabigoon River system, Rudd et al. 1983; the Ottawa River, Miller and LaHam 1977) which have established that increased loading of inorganic mercury to lakes and rivers can increase the methyl mercury concentrations in fish. As the radioisotope additions represented a significant increase in dissolved Hg concentrations in most of our experiments (Table D5), some experiments were done to determine what effect dissolved Hg additions had on results.

In 1984, mercuric chloride additions were made at three different added mercury concentrations which would have increased the total Hg concentrations in the limnocorrals by 0, 2 (the concentration created by a radioisotope spike in 1984) and 10 $ng \cdot L^{-1}$. At the end of August (at which date experiments in 1981-1983 were terminated), these three limnocorrals were indistinguishable in their total Hg concentrations in yellow perch muscle (Fig. D10), but by the end of September the limnocorral with highest addition had a concentration in yellow perch muscle of 0.094 $\mu g \cdot g^{-1}$ while the zero addition limnocorral and radioisotope spiked limnocorral had concentrations of 0.069 and 0.058 $\mu g \cdot g^{-1}$ respectively. These results indicated that, over the normal course of most experiments, approximately July 1 to August 30, a single pulse addition raising the Hg concentration in the limnocorrals by as much as 10 $ng \cdot L^{-1}$ would not affect the outcome of the experiments. This pulse concentration of Hg was higher than that created in any year by the radioisotope spiking (highest 8 $ng \cdot L^{-1}$ in 1981, Table D5). The corroborative experiment in 1984 was the addition of moss-peat material to two limnocorrals only one of which received a radioisotope spike. By the end of August the mercury concentration in yellow perch muscle was identical between these two limnocorrals at 0.14 $\mu g \cdot g^{-1}$ (Fig. D10), but by the end of September the spiked limnocorral had a higher fish mercury concentration (0.16 $\mu g \cdot g^{-1}$ vs 0.12 $\mu g \cdot g^{-1}$).

Over longer time periods and with multiple additions, the mercury added with the radioisotope spike did affect the limnocorrals. As seen above, as early as September of the year of the spike, a 10 $ng \cdot L^{-1}$ concentration pulse in an untreated control limnocorral or a 2 $ng \cdot L^{-1}$ pulse in a moss-peat limnocorral began to increase mercury in the fish. In 1982 and 1983 limnocorrals originally spiked the previous year were rerun (Table D1). In both years these rerun limnocorrals, which had received two spikes of mercuric chloride, exhibited higher microbial methylation rates (as measured by Ramlal et al. 1986) than new limnocorrals. This effect was evident in both controls and moss-peat treated limnocorrals. In 1983, a rerun control limnocorral developed the higher fish total mercury concentration (1.1 $\mu g \cdot g^{-1}$) of all the experiments conducted in all years (Appendix Table DA3.39). This limnocorral also had a high microbial methylation rate (Ramlal et al. 1986).

Clearly, multiple isotope additions can create mercury contaminated ecosystems which are not comparable with new limnocorrals. We had intended to follow the effect of aging on some of the treatments by reusing these treated limnocorrals in successive years; however, this approach had to be abandoned. It might be done on limnocorrals which did not have the radioisotope added, but the isotope offered other advantages to be discussed below. There was no evidence that the radioisotope spike altered the behavior of control or treatment limnocorrals over the period of most experiments, 1 July through end of August. In fact, fish mercury concentrations in control limnocorrals in all years generally remained unchanged or declined over the course of the experiment (Fig. D13b, Table D9). Therefore, limnocorral experiments were testing only the effect of the terrestrial materials added, and the results were independent of the mercury added with the spike during the first season of limnocorral use.

The radioactive mercury measured in fish muscle is almost entirely methyl mercury by the end of the experimental period (Table D8). In controls methyl mercury was 79 to 92% of total radioactive mercury while in treated limnocorrals essentially all the radioactive Hg was in the methyl form. This proportion increased with time with the control fish increasing more slowly than the treated fish. These results are consistent with the general finding that methyl mercury account for around 90% of the total mercury in fish muscle

(Huckabee et al. 1979). Consequently, when experimental treatments resulted in increased mercury concentration in fish muscle, these increases were due to increased methyl mercury concentrations caused by the treatment.

The results of experiments were independent of fish species. In 1981 emerald shiners and yellow perch were both added to the limnocorrals. In addition, extraneous species were present in control and treated limnocorrals. There was excellent correspondence between the ^{203}Hg activity of perch, shiners and other species of comparably sized fish. Because all these species had similar diets, aquatic invertebrates, it is not possible to say how alternative diets, especially piscivory, would have affected results. In natural systems, piscivores usually have higher mercury concentrations in their muscle than benthivores or planktivores (Huckabee et al. 1979), and this is true of fish from lakes in the Churchill River diversion (Bodaly et al. 1984a; Hecky et al. 1987; Derksen 1987).

MOSS-PEAT TREATMENTS

Moss and peat are the dominant ground cover and organic soil in the low-lying areas of northern Manitoba which were inundated by the new impoundments of the Lake Winnipeg, Churchill and Nelson Rivers study. These materials may account for up to one-third of the total amount of terrestrial organic material (excluding tree boles) added to new reservoirs from flooded land (G.K. McCullough, unpublished data), and this material is likely intermediate in its susceptibility to bacterial decomposition in reservoirs between highly degradable herbs and tree leaves and the tree boles which break down very slowly. Moss and peat material has the advantage to the limnocorral experiments of being relatively easy to harvest and handle quantitatively. Consequently, it was the terrestrial organic material experimented with most often as new limnocorrals were treated in every year (Tables D1 and D2).

A moss-peat treatment of 400-500 kg (wet weight) per limnocorral in 1982, 1983 and 1984 approximately doubled the mercury concentration in yellow perch muscle relative to controls within two months (Figs. D3, D10, D13a and D13b). The quantitative effect of the moss-peat material was highly replicable in two limnocorrals in 1983 (Fig. D7) as was the qualitative effect in all years. However, in 1981 the moss-peat effect was even more marked, with the treated limnocorral fish having nearly four times the mercury content of other years (Fig. D13a). The effect of moss-peat material in 1982 was proportional to the weight of the material added with a doubling of weight doubling the mercury incorporated into yellow perch muscle (Fig. D3). In 1982, a limnocorral was installed over a terrestrial surface with a moss and sedge community that had been flooded in 1976 with the beginning of Southern Indian Lake impoundment. Fish mercury in this limnocorral, which contained material inundated for six years, was indistinguishable from the controls (Fig. D3). Other attempts to look at the effect of aging on moss-peat material were confounded by the mercury contamination of the material caused by multiple mercury spikes as discussed above.

BLACK SPRUCE

Black spruce is the overwhelmingly dominant tree in northern Manitoba and constitutes the majority of organic biomass added to northern reservoirs (G.K. McCullough, pers. comm.). Although the boles can be extremely resistant to decay in the water, the needles and the finer twigs can break dowm rapidly (Crawford and Rosenberg 1984). The rapid breakdown of this material caused the limnocorral receiving it in 1983 to go anoxic (Table DA1.3) at which time the fish died. In 1984, the weight of spruce added was correctly scaled to the Notigi Reservoir by reducing it to 1/3 the 1983 addition (Table D2) and the experiment was successfully completed. By the end of August and at the end of the experiment in September, black spruce was more effective than the moss-peat material in stimulating mercury accumulation in yellow perch muscle (Fig. D10).

PRAIRIE SOD

This treatment is not relevant to the mercury problem in Churchill River diversion, but it was conducted (solely funded by Department of Fisheries and Oceans) to determine if the stimulatory effect of flooding on fish mercury would be a general response or whether it might only be a problem in the boreal forest. The sod treatment caused higher mercury concentrations in fish relative to controls (Fig. D10), but it also caused more rapid fish growth (Fig. D9). Consequently, the final Hg concentrations in fish muscle in the sod limnocorral were lower than the moss-peat and spruce treatments. Fish from the sod and spruce limnocorrals did have approximately the same muscle burden, i.e. µg Hg per muscle fillet, indicating that uptake of methyl mercury was essentially the same in all three organic treatments in 1984. Only the growth of fish differed among the organically treated limnocorrals.

INORGANIC NUTRIENTS

Inundation can release inorganic nutrients from vegetation and soil and thereby may increase phytoplankton primary production. This stimulation was observed in some parts of Southern Indian Lake (Hecky and Guildford 1984) and in Notigi Reservoir (Bodaly et al. 1984b; Jackson and Hecky 1980). In the limnocorral experiments of Rudd and Turner (1983b), increased primary production increased organic loading which stimulated bacterial growth and methylation of mercury as well as fish growth. The phosphorus and nitrogen additions made to a limnocorral in 1981 boosted chlorophyll concentrations and primary productivities (Guildford 1984) to levels comparable to Notigi Reservoir during and since impoundment (Jackson and Hecky 1980; Bodaly et al. 1984b; Bodaly et al. 1987). The rate of primary productivity in the treated limnocorrals was double the rate of productivity in control limnocorrals, yet there was no effect on the uptake of ^{203}Hg (Table D7; Fig. D1), total Hg uptake (Table D9) or fish growth (Appendix DA3). It should be noted that Rudd and Turner (1983b) boosted primary productivity in their limnocorrals by factors of 4 and 9 times over controls, i.e. to a much greater extent than our experiments. Although the inorganic nutrient addition increased organic loading by primary production from approximately 2.3 kg organic carbon to 4.5 kg organic carbon per limnocorral over the course of the experiment, the moss-peat and spruce treatments added 50 and 30 kg organic carbon respectively to each enclosure. Therefore, the potential effect of the moss-peat and spruce treatments on bacterial growth and mercury methylation was much greater over the two month experimental period because of the greater availability of organic substrate.

SEDIMENT EFFECTS

Southern Indian Lake has had high rates of shoreline erosion since it was flooded in 1976 (Newbury and McCullough 1984). This erosion has altered the sediment budget of the lake by greatly increasing suspended sediment concentrations and sedimentation rates (Hecky and McCullough 1984). Notigi Reservoir has not suffered the same high erosion rates and suspended sediment concentrations are low throughout much of the reservoir except near where the diversion channel enters from Southern Indian Lake. As eroded suspended sediments represent a new source of mercury to the reservoirs, a major difference in the water quality of Southern Indian Lake and Notigi Reservoir, and a possible mitigation measure (Rudd et al. 1983), an extensive series of experimental additions of suspended sediments were done in 1981-83 (Tables D1 and D2).

All the experiments indicated that suspended sediments had either a negligible or a stimulatory effect on mercury accumulation in fish. In plastic bottomed limnocorrals in 1981, the lowest rate of suspended sediment addition (eroded bank clay) led to somewhat higher rates of ^{203}Hg uptake in fish compared to controls (Table D7), but this effect was not apparent in the total mercury analyses (Table D9). The uptake of mercury by fish in the limnocorral with the higher rate of sediment addition was not distinguishable from controls. In 1982 a natural lake bottom was treated at a rate of sediment addition similar to the high rate in 1981. Over much of the experiment, this treatment was identical to the controls in its fish mercury content, but then the fish mercury increased rapidly so that mercury concentrations at the end of August were similar to or higher than the moss-peat treatments (Fig. D3). There was no evidence in these experiments of depression of mercury accumulation by suspended sediment originating from eroding clay banks in Southern Indian Lake.

The addition of suspended sediments has been suggested as a possible mitigation measure in the mercury contaminated English-Wabigoon system (Rudd and Turner 1983a). To test the short term efficacy of such a measure in the Churchill River diversion area, suspended sediment was added to a moss-peat treated limnocorral in 1983. As the eroded bank clay had already been shown not to depress mercury accumulation in fish, lake sediments from Wupaw Bay were used in the 1983 experiment. For mitigation, resuspension of lake sediments by dredging would be the most feasible means of altering suspended sediment concentrations. The effect of suspended lake sediment on total mercury uptake by fish was compared to two other moss-peat limnocorrals (Fig. D7). There was no effect of the suspended sediments on total mercury uptake. The uptake of ^{203}Hg was enhanced by the sediment addition (Fig. D6) suggesting that more spike mercury was made available for methylation by the suspended sediment treatment. In any case, there was no evidence that suspended sediment addition had any mitigatory value. In the longer term, covering flooded terrestrial material with a layer of sediment may be effective; and this may explain why the six year old, undisturbed, flooded land surface in the 1982 experiment did not stimulate mercury accumulation in fish (Fig. D3).

SOURCES OF MERCURY

The addition of ^{203}Hg and cold (non-radioactive) mercury with the radioisotope spike created a pool of tagged mercuric ion which could then be followed through the contained ecosystem. Between 30 and 50% of this added mercuric ion rapidly, within one day, became associated with particulate material, and the proportion of particulate to total ^{203}Hg in the water column was relatively stable over the life of the experiments (Appendix DA2). Because of this affinity for particles, the added mercury was rapidly removed from the water column by sedimentation. Typical times for removal from the water column of 50% of the added mercury were 8 to 15 days (Appendix DA2). Variation among limnocorrals in removal times and partitioning between dissolved and particular forms did not explain the observed patterns in fish muscle mercury. Similarly, the ^{203}Hg activity in the food of yellow perch, net plankton and benthic organisms (Appendix DA4), was not related to either ^{203}Hg or total Hg content of the fish. As the ^{203}Hg traces the total mercury pool created by the spike and only methyl mercury accumulates in fish muscle (Table D8), the lack of correlation between fish mercury and biotic or abiotic forms of total mercury is not surprising if methyl mercury was a small fraction of total mercury in these other forms. Methyl mercury in water rarely exceeds 10% of total Hg in natural waters and is usually much less (e.g. <0.1% in Clay Lake, Ontario, Rudd et al. 1983). Within the biota, only fish selectively accumulate methyl mercury. Methyl mercury in zooplankton samples from many lakes and reservoirs in northern Manitoba was undetectable while total mercury concentrations ranged from 100 to 700 ng$\cdot$g$^-$ dry weight (Ramsey and Ramlal 1985).

The mercury added with the radioisotope spike was a relatively small portion of the total mercury taken up by fish in the organic treatments. Before the radioisotope is added, there is no radioisotope within the mercury "pools" in the limnocorral represented diagrammatically in Fig. 1. These pools become labelled over time. Some of the pools label rapidly and come into radioisotopic equilibrium which is maintained throughout the experiment, e.g. the proportion of dissolved and suspended ^{203}Hg mentioned above and the proportions of dissolved fractions measured in 1981 (Appendix DA2). However, the fish come to equilibrium slowly with the spike because the concentration of methyl ^{203}Hg in fish muscle is dependent on an uptake rate of methyl Hg from water and food which in turn is dependent on how rapidly the dissolved methyl mercury pool is increased and labelled by bacterial methylation. After the addition of the radioisotope, the specific activity of the mercury, i.e. the proportion of radioactive Hg atoms to the total number of mercury atoms in fish muscle, increases over time until equilibrium with the various sources of mercury is attained. At equilibrium, it is possible to compare the specific activity of the spike with that of the equilibrated mercury pool and determine how much of the mercury in the pool is from non-spike sources. Because cold (non-radioactive) mercury concentrations in the water column are immeasurable, only the specific activity of the mercury in fish muscle can be determined over time.

There were five experiments in which nearly constant specific activities were achieved while mercury body burdens in fish were still increasing. These conditions indicate that the methyl mercury pool was in equilibrium with the sources of inorganic mercury including the spike. From these data the size of the mercury pool available for methylation in these experiments can be estimated (Table 1). Spike mercury was never more than one-third of the total mercuric ion pool. The moss-peat treatment made the most mercuric ion available, 7.4 mg and 8.9 mg in new limnocorrals in 1981 and 1984. A year of submersion of the moss-peat in the limnocorral apparently reduced the available mercury pool from 7.4 in 1981 to 6.4 a year later, but this change may not be significant as it was less than the differences observed between two new limnocorrals. The sediment and spruce additions made much less mercuric ion available. In all cases, the available mercuric ion pool was a very small proportion, always less than 0.5% of the total mercury added in the treatment (Table 1).

The amount of mercury added in the spike, the concentration of mercury in treatment materials, and the mercuric ion pool size were not related to the total mercury concentration in fish muscle at the end of the experiments. For example, in 1984, the spruce treatment had the lowest concentration of mercury, the lowest amount of mercury added with the treatment and the smallest mercuric ion pool, yet it had the highest mercury concentration in fish muscle of all treatments. Mercury availability in the limnocorrals did not appear to be the sole factor or even a demonstrably important factor in the mobilization and accumulation of mercury in fish muscle. However, multiple mercury spikes did affect limnocorrals over time. Reused limnocorrals always had higher methylation to demethylation ratios than new limnocorrals (Ramlal et al. 1986). Multiple mercury spikes which expose bacterial populations to elevated mercuric ion concentrations for extended periods of time may exert a strong selective pressure for bacteria which can detoxify their cellular environment by producing methyl mercury. Once selected for, these bacteria may more efficiently methylate mercuric ion in limnocorrals.

BACTERIAL METHYLATION

Bacteria play a central role in the aquatic mercury cycle of the limnocorrals (Fig. 1) as they can both methylate and demethylate mercury species. Methylation can be carried out by a broad spectrum of bacteria under anaerobic and aerobic conditions as a result of incorrect synthesis of essential biochemical compounds (Wood et al. 1972). In contrast, demethylation is performed only by bacteria which have a specific enzyme, mercuric reductase. For methyl mercury concentrations to increase, rates of methylation must exceed demethylation for a significant period of time. The methyl mercury produced can accumulate rapidly in fish. Because of its high solubility in lipids, methyl mercury passes readily through cell membranes and because of its strong affinity for sulfhydryl groups it becomes strongly bound to the proteins in fish muscle. Fish can depurate mercury, but this process is extremely slow with a biological half-life (time to eliminate 50% of mercury content from a biological organism) in excess of two years (Huckabee et al. 1979). Consequently, fish muscle acts as an efficient accumulator of methyl mercury over the two to three month period of our experiments. The increases in mercury in fish were the result of the changing methyl mercury concentrations in the limnocorrals, which in turn was controlled by bacteria.

All of the treatments in which terrestrial organic material was added to limnocorrals resulted in higher mercury concentrations in fish relative to controls. As discussed above, these increases were not caused by increasing the total mercury or the mercuric ion pools in the limnocorrals at least over the first few months. In the longer term, adding inorganic mercury certainly can affect fish mercury. The dynamics of the response appears to depend on bacterial activity. Organic materials are known to stimulate bacterial growth and bacterial methylation (Fagerstrom and Jernelov 1972; Wright and Hamilton 1982; Furutani and Rudd 1980), and the qualitative response of organically loaded limnocorrals is consistent with earlier studies. Adding mercury alone to limnocorrals does not stimulate bacterial growth, but it can select for mercury resistant bacteria which are efficient methylators. The speed of the selection process may depend on the number of generations of bacteria produced, i.e. growth rate. The addition of organics should speed the selection process which shifts the methylation-demethylation balance in addition to maintaining higher bacterial growth rates once the selection process is completed.

Ramsey and Ramlal (1985, 1986) have shown that bacterial methylation rates increase rapidly in August in reservoirs and natural lakes in northern Manitoba with the reservoirs eventually achieving rates, on average, three to five times higher than unflooded lakes. Limnocorral results also indicate that August is the period of rapid uptake of methyl mercury by fish. For example, in 1984 when fish in limnocorrals were sampled weekly, fish in organically treated limnocorrals achieved 70-80% of their overall change in mercury concentration between August 3 and August 25 (Fig. D10). Control fish also showed a slight increase in their mercury concentration at the same time. The 1984 time course of mercury uptake in limnocorrals corroborates the observations of Ramsey and Ramlal and further substantiates the critical role of bacterial methylation in fish mercury uptake.

CONCLUSION

Moss-peat and spruce, the dominant soil and vegetation components of the boreal forest, stimulate bacterial methylation and cause increased methyl mercury concentrations in fish muscle. At least in the case of moss-peat, the effect on fish mercury is dose dependent. Given these observations and the relative uniformity of the flooded terrestrial community in the Churchill River diversion area, the effect of impoundment on fish mercury should be proportional to flooded area as previously suggested by Bodaly et al. (1984a).

Mercury concentrations in flooded materials, the total mass of mercury in these materials and the amount of mercury available for methylation did not predict the outcome of experiments within one season. However, spiking limnocorrals with inorganic mercury did alter the methylation-demethylation of limnocorrals after one year. The amount of mercuric ion that the added organics supplied was large relative to the radioisotope spike mercury but extremely small relative to the amount of total mercury added with the organic and sediment treatments. Mercury in flooded materials enter the mercuric ion pool slowly, probably dependent on the decomposition of the organic materials; and this implies that many years may be required for the flooded Hg to pass through the aquatic cycle to eventual burial.

In the limnocorral experiments, eroded bank clay added in suspension was either neutral or stimulatory to bioaccumulation. Lake sediments added in suspension over flooded moss also did not reduce bioaccumulation of Hg. Based on these experimental results, increasing concentrations of suspended sediment, for example by dredging, cannot be recommended as a mitigatory action in the Churchill River diversion. Also, the high suspended sediment concentrations which occur in Southern Indian Lake have likely not been a factor in that lake's fish having lower mercury concentrations than Notigi Reservoir.

High suspended sediment concentrations should be associated with high sedimentation rates, and rapid burial of flooded organic materials may hasten the recovery of new reservoirs from high mercury concentrations in fish. Although this has not been specifically investigated in the limnocorrals, the negligible uptake of mercury in a limnocorral installed over a six year old land surface which had a thin covering of sediment would be consistent with the mitigatory effect of prolonged or rapid sedimentation. The results in the six year old land surface are inconclusive in this regard because of differences in the plant community and the manner of addition. In any event, as much of the flooded material in new reservoirs occurs in the shallowest depths while the highest rates of sedimentation in the long term generally occur in the deepest depths, sedimentation may not be effective in reducing the stimulatory effect of organic materials on Hg accumulation in fish.

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Table 1. Experiments in which the specific activity of fish muscle mercury came to equilibrium with mercury sources. Limnocorral designators (from Hecky et al. 1986), treatment, total mercury added with treatment (1), mercuric ion added with spike (2), specific activity (SA) of spike (3), specific activity (SA) of fish at end of experiment (4) and calculated pool size of available mercuric ion in the limnocorral. Calculation for available Hg is ((3)÷(4)x(2)=(5)).

Limnocorral	Treatment (dry weight)	Total Hg in treatment	Total Hg added in spike	Spike S.A.	Fish S.A.	Available treatment Hg
		mg (1)	mg (2)	10^8 Bq(mg Hg)$^{-1}$ (3)	(4)	mg (5)
LM81-2	63 kg moss-peat	6300	1.82	0.85	0.210	7.4
HS82-7	50 kg bank clay	2000	1.14	1.22	0.583	2.4
SP84-3	31 kg spruce	900	0.66	2.11	0.359	2.1
SD84-4	610 kg sod	36600	0.66	2.11	0.293	4.8
LM84-5	53 kg moss-peat	2100	0.66	2.11	0.156	8.9
OM82-5	63 kg moss (+ 1 year)	6300	1.14	1.22	0.218	6.4

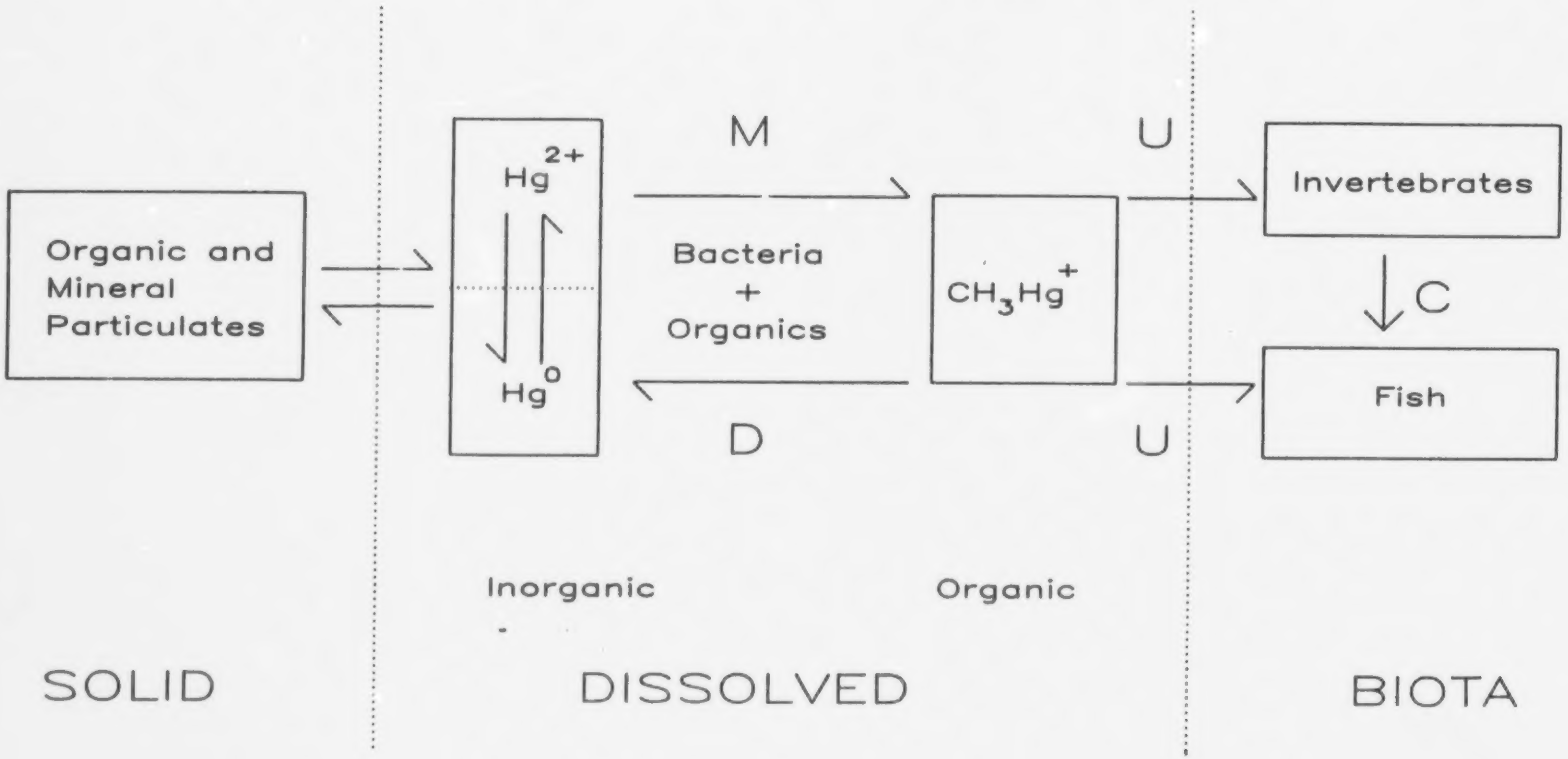
Organic and Mineral Particulates
Hg^{2+}
Hg^{0}
M
Bacteria + Organics
D
CH_3Hg^+
U
U
Invertebrates
C
Fish
Inorganic
Organic
SOLID
DISSOLVED
BIOTA

Fig. 1. The aquatic Hg cycle.

Canadian Data Report of
Fisheries and Aquatic Sciences 628

March 1987

MERCURY BIOACCUMULATION IN YELLOW PERCH IN LIMNOCORRALS SIMULATING THE EFFECTS OF RESERVOIR FORMATION

by

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This is the 8th Data Report
from the Central and Arctic Region, Winnipeg

SPECIAL NOTE

This report was prepared for the Canada-Manitoba Agreement on the Study and Monitoring of Mercury in the Churchill River Diversion. Any conclusions, recommendations, or opinions expressed herein are those of the author and are not necessarily the same as those of the Steering Committee for this Agreement.

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Cat. no. Fs 97-13/628E ISSN 0706-6465

Correct citation for this publication is:

Hecky, R.E., R.A. Bodaly, N.E. Strange, D.J. Ramsey, C. Anema, and R.J.P. Fudge. 1987. Mercury bioaccumulation in yellow perch in limnocorrals simulating the effects of reservoir formation. Can. Data Rep. Fish. Aquat. Sci. 628: v + 158 p.

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ABSTRACT

Hecky, R.E., R.A. Bodaly, N.E. Strange, D.J. Ramsey, C. Anema, and R.J.P. Fudge. 1987. Mercury bioaccumulation in yellow perch in limnocorrals simulating the effects of reservoir formation. Can. Data Rep. Fish. Aquat. Sci. 628: v + 158 p.

Four series of limnocorral experiments were conducted to investigate how reservoir formation could increase mercury concentrations in fish. The limnocorrals were 10 m diameter and 2-4 m in depth. They were stocked with young yellow perch and maintained from 60 to 90 days. Experimental treatments included organic materials (moss-peat, spruce boughs, and prairie sod), suspended sediments, inorganic nutrients (to stimulate primary production) and inorganic mercury. Loadings of these materials were made, with the exception of inorganic mercury, to match the water volume:treatment material ratio of known reservoirs.

The ^{203}Hg isotope of mercury was added in order to estimate the available mercury in the experimental systems. The uptake of ^{203}Hg and total Hg by yellow perch was monitored at regular intervals over the course of the experiments. The organic materials consistently stimulated bioaccumulation of methyl mercury in perch muscle. Suspended sediments were generally neutral or stimulatory to Hg bioaccumulation. Inorganic nutrients were not stimulatory. Inorganic mercury did not enhance bioaccumulation within 90 days of its addition. Most of the mercury bioaccumulated in limnocorral fish was natural mercury as mercury added with the radioactive spike never accounted for more than 17% of the mercury in the fish at the end of the experiments.

Key words: mercury; fish; reservoirs (water); limnocorrals; methyl mercury; bioaccumulation; resuspended sediments; soils; vegetation cover.

RÉSUMÉ

Hecky, R.E., R.A. Bodaly, N.E. Strange, D.J. Ramsey, C. Anema, and R.J.P. Fudge. 1987. Mercury bioaccumulation in yellow perch in limnocorrals simulating the effects of reservoir formation. Can. Data Rep. Fish. Aquat. Sci. 628: v + 158 p.

Quatre séries d'expériences en enceintes lacustres ont été effectuées afin d'établir comment la formation des réservoirs pouvait donner lieu à des concentrations en mercure plus élevée dans les poissons. Le diamètre des enceintes lacustres était de 10 m et la profondeur de l'eau, de 2 à 4 m. On y a mis de jeunes perchaudes, qui y sont restées de 60 à 90 jours. Des traitements expérimentaux ont alors été faits au moyen de matières organiques (tourbe mousseuse, rameaux d'épinette, gazon de prairie) de sédiments en suspension, de substances nutritives inorganiques (afin de stimuler la production primaire) et de mercure inorganique. Ces substances, à l'exception du mercure inorganique, ont été ajoutées en quantité telles que le rapport volume d'eau/substances pour le traitement correspondait à celui de réservoirs connus.

L'isotope ^{203}Hg a été ajouté afin d'estimer la quantité de mercure dans les systèmes expérimentaux. La fixation de ^{203}Hg et la concentration totale en Hg dans les perchaudes ontété contrôlées à intervalles réguliers pendant la réalisation de l'expérience. Les substances organiques stimulaient toujours la bioaccumulation de mercure dans les muscles de la perchaude. Les sédiments en suspension, de façon générale, étaient neutres ou stimulaient la bioaccumulation de Hg, contrairement aux substances nutritives inorganiques. Le mercure inorganique n'a pas favorisé la bioaccumulation dans les 90 jours qui ont suivi son addition. Le mercure naturel représentait la plus grande partie du mercure accumulé dans les tissus des poissons dans les enceintes lacustres le mercure légèrement radioactif qui avait été ajouté ne comptant jamais pour plus de 17% du mercure dans les poissons à la fin de l'expérience.

Mots-clés: mercure; reservoir (eau de); enceintes lacustres; mercure méthyle; bioaccumulation; sédiments remis en suspension; sols; couverture végétale.

INTRODUCTION

Limnocorral experiments on mercury bioaccumulation in fish were conducted at Southern Indian Lake in northern Manitoba in 1981-84 to test the effect of treatments which simulated conditions in new northern reservoirs. Bodaly et al. (1984a) documented the increase in mercury concentrations in Southern Indian Lake, Notigi Reservoir and other lakes flooded by the Churchill River Diversion (see Newbury et al. 1984 for details of the diversion project). They also cited numerous other instances of elevated fish mercury concentrations in reservoirs. Mercury methylation in aquatic environments is primarily bacterially mediated (Bisogni 1979). Bodaly et al. (1984a) hypothesized that the addition of new organic substrate from flooded shorelines would enhance bacterial growth and the methylation of mercury. The limnological conditions in Southern Indian Lake and Notigi reservoir during and after impoundment were known (Hecky et al. 1984; Bodaly et al. 1984b), and this knowledge permitted realistic simulation experiments to be conducted and relevant hypotheses to be tested.

A limnocorral approach offered the best compromise between control and realism. In every year either controls or a treatment were replicated to estimate the variance between limnocorrals. A radioisotope of mercury was added as an inorganic salt ($HgCl_2$) to provide sensitivity to our analysis. The bivalent mercury ion is the only species of mercury which is methylated (Bisogni 1979) so that by tagging this pool the methylation of mercury and the movement of mercury through the food chain can be followed. Total dissolved mercury concentrations were always below or at the detection limit in the Churchill River Diversion area (Bodaly et al. 1984a, 1987). Consequently the movement of mercury from the water to the biota could not be followed without the radioisotope. Significant amounts of mercury are added with the radioisotope spike and ambient mercury concentrations are unknown, so the experiments are not true tracer experiments. However, the radioisotope does provide a means of following the bioaccumulation of available mercury into fish and comparing the effects of different treatments. This approach and methodology was chosen because it was successfully applied to the mercury problem in the English-Wabigoon river system in northwestern Ontario (Rudd et al. 1983).

In each year of limnocorral experiments, the effects of different terrestrial materials on mercury accumulation in fish were investigated. In 1981 moss and underlying peat which forms the dominant ground cover and fine-grained clays which form the greatest mass of material eroding into Southern Indian Lake (Newbury and McCullough 1984; Hecky and McCullough 1984) were used as treatments. In 1982 these same materials were again used but emphasis in the experiments was on different loading rates of moss-peat, the effect of time of submersion of the moss-peat and the effect of eroded clays on the methylating activity of lake sediment. In 1983, the reproducibility of the moss-peat treatment was examined as well as the effect of time of submersion and the combined effect of moss-peat and suspended sediment added together. In 1984, the last year of experiments, different kinds of organic substrates, moss-peat, black spruce and prairie soil were studied as well as the effect of mercury concentration alone.

METHODS

LIMNOCORRAL SITE

The limnocorrals were all installed in a small protected bay off Wupaw Bay, Southern Indian Lake. Bodaly and Lesack (1984) concluded that reservoir formation had minimal effect on northern pike in the bay. Wupaw Bay was inundated in 1976, but it had the lowest proportion of flooding of any region on Southern Indian Lake (only 9% increase in surface area). Yellow perch in the bay have the lowest concentration of Hg of any region surveyed in Southern Indian Lake, but concentrations in perch were somewhat higher than perch from unflooded control lakes (Bodaly et al. 1987). The protected nature of the bay combined with the large amount of background information for the area made it the preferred choice for siting the limnocorral experiments.

LIMNOCORRAL CONSTRUCTION

The limnocorrals used in these experiments were installed in 10 m diameter wooden rings floating on styrofoam blocks attached with plastic strapping. The wall material of the enclosures was translucent, fiber-reinforced polyethylene. In 1981 five of the limnocorrals had sewn-in plastic bottoms (Table 1) while two had their walls sealed to the lake sediment surface (a silty clay) with sand bags placed by SCUBA divers on the excess wall material. In all other years all limnocorrals were sealed to the sediment surface. All the limnocorrals in a given year were installed between two and four meters deep within a period of two to three days in late June.

A known amount of tritiated water was added to each limnocorral after installation. The observed dilution of the spike of tritium allowed estimation of the volume of water in each limnocorral (Table 1). From this volume estimate, the mean depth of the limnocorral could be calculated assuming a 10 m diameter. Monitoring of tritium activity throughout the duration of the experiments served as a check for leakage. No leakage problems were observed in newly installed limnocorrals over the 60-90 day course of these experiments. Some limnocorrals were reused the following year after having been left installed over the winter. In one instance, some leakage occurred in one of these reused limnocorrals (Table 1; OC82-5), but this was easily corrected by sewing the material.

TREATMENTS

The treatment materials added to each limnocorral are identified in Table 1 and specific details are given in Table 2. In every year at least one control limnocorral was established which received no other treatment after installation other than the radioisotope spike and the addition of fish (see below). Also in every year at least one limnocorral received a "low" moss-peat treatment. A "low" treatment was defined as the addition of approximately 500 kg wet weight of moss-peat grubbed from the soil surface to the depth of permafrost (normally about 10 cm in mid-June). In every year this material was taken from an area immediately adjacent to the bay in which the limnocorrals were installed. In 1982 the "high" moss-peat treatment consisted of the addition of 1000 kg wet weight of moss-peat material. The low moss-peat treatment was scaled to the ratio of water volume to flooded area of Notigi Reservoir and the flooding of a moss-peat layer of mean thickness of 5 cm. Notigi Reservoir recorded the highest fish mercury concentrations of any lake in the Churchill River Diversion area (Bodaly et al. 1984a). The moss-peat was grubbed by hand and added as a single addition prior to the addition of fish and radioisotope spikes. Most of the material sank soon after addition; however a portion would generally float for up to a week before sinking.

Sediment treatments in 1981 and 1982 used varved, extremely fine-grained, glaciolacustrine clays from erosion site 5 of Newbury and McCullough (1984). In 1983 lake sediment from a nearby lake station, LBN of Planas and Hecky (1984), was used. The bank material or lake sediment was collected by shovel or Ekman dredge as appropriate and placed in large plastic trash cans. Water from the limnocorral to be treated was added to the plastic can with a plastic vaned submersible pump. After agitation to bring the clay material into suspension, the water was pumped back into the limnocorral and dispersed in the prop wash of an electric outboard motor. Turbid water was thus added until a predetermined Secchi disc value was reached in the limnocorral (0.3 and 0.5 m for high and low treatments in 1981, 0.35 m in 1982 and 0.5 m in 1983). At the end of the experiment the residual clay in the trash can was recovered, dried and weighed to calculate the total material added (Table 2).

In 1981 inorganic nutrients, $NaNO_3$ and $NaH_2PO_4 \cdot 2H_2O$ were added in order to stimulate algal growth and primary productivity. Three to four grams of $NaH_2PO_4 \cdot 2H_2O$ were added biweekly from June 21 to August 21. Twenty-five grams of $NaNO_3$ were added on June 21 and July 15 only. The nutrient salts were dissolved in 400 mL double distilled water then mixed into the limnocorral in the prop wash of an electric outboard motor.

In 1982 a limnocorral was installed over terrain which was flooded when Southern Indian Lake was impounded in 1976. This limnocorral was reused in 1983. The site was selected from pre-impoundment aerial photographs which indicated a low-lying boggy area. The presence of moss-peat material as well as a high proportion of grass and sedge was confirmed by SCUBA diver prior to installation. In 1982 the surface of this flooded terrain was not disturbed after installation. In 1983, prior to the radioisotope spike, the surface was purposely disrupted by SCUBA divers using their hands and mechanical aids to expose underlying materials.

In 1984 the spruce and sod treatments were scaled on the flooded area to volume ratio of Notigi Reservoir. Sod was from an abandoned farm field on the Canadian Department of Agriculture station at Glenlea, Manitoba. A motor driven sod cutter was used to produce uniform pieces approximately 30 cm x 60 cm x 5 cm. The treatment was made on an areal basis, but no attempt was made to insure that sod was oriented grass side up. The weight of spruce boughs (less than 2 cm in diameter) to be added was based on vegetation surveys conducted around Wupaw Bay in 1976 (J.H. Shay, University of Manitoba, Winnipeg, MB, unpublished data) which estimated biomass of spruce components per square meter of land.

The carbon, nitrogen, phosphorus and total mercury concentrations were determined on all added materials. After determination of water content by weight loss at 105 C, subsamples were ground with either mortar and pestle or a mechanical ball mill. The total C, total N, total P and carbonate content of the treatment materials were determined by the methods of Stainton et al. (1977). Digestion of sediment and soil for total Hg was carried out following the aqua regia procedure of Dow Chemical Co. of Canada Ltd. (Method CAS-AM-70.13). The mercury concentration was determined by the atomic absorption spectrophotometric method of Armstrong and Uthe (1971). To determine the total amount of these elements added to the limnocorrals the concentration per unit dry weight was multiplied by the calculated dry weight of material added.

FISH ADDITIONS

Before the addition of experimental fish, fish trapped incidentally during installation were removed by fishing with an 18 m seine net, variable mesh gill nets, minnow traps and by electroshocking. Despite these removal efforts, some non-stocked fish, generally young of small species, were recovered in every year at the end of the experiments. Limnocorrals were then stocked with fish seined from nearby areas in Wupaw Bay. Stocking densities per enclosure were nominally as follows: 110 perch and 30 emerald shiners in 1981, 80 perch in 1982, 32 perch in 1983 and 100 perch in 1984 (Table 3). In 1981, fish were were held briefly in pens for one to two days before the limnocorrals were stocked, whereas in 1982-84 fish were added immediately after transport from seining areas. Yellow perch of the same age class (1+) were used in experiment in all years except 1983, when perch of the 1+ class were difficult to find in Wupaw Bay and 2+ were initially stocked. Also, 60 age 0+ perch were added in 1983 (Table 3) on August 5. In 1981 only, 2+ emerald shiners were also stocked.

A sample of the fish introduced to the limnocorrals, the "time zero" sample, was removed during stocking. Fish were sampled at two week intervals using the 18 m seine net, minnow traps or an electroshocker as required. In 1983 sampling was less frequent due to lower initial population in the enclosures. Larger, more frequent samples were taken in 1984 when initial populations were high and survivorship was good. For the final sample each year, rotenone was added to the limnocorrals and all remaining fish were dipnetted from the surface.

All fish sampled were placed immediately on ice. At the South Indian Lake research station, fish were individually wrapped and frozen. Table 3 summarizes the fish stocked to each limnocorral, the numbers of stocked fish recovered over the experiment and the total (including extraneous fish) recovered. Fish growth in each year is summarized in Table 4.

RADIOISOTOPE ADDITION

In 1981 a cocktail consisting of radioisotopes of four metals in addition to ^{203}Hg was added to each limnocorral (Tables 5 and 6). The radioisotopes ^{203}Hg, ^{134}Cs, ^{65}Zn and ^{60}Co were added as their chloride salts while ^{75}Se was added as seleneous acid. These compounds in 5 mL of 0.5 M HCl solution were added directly to the limnocorrals from their containing glass vial. In addition in 1981 a radioisotope of Na was added to one of the limnocorrals (Table 6). In all other years only radiomercury was added to limnocorrals. In all years tritiated water was added to check for leakage and to estimate volumes. Table 5 gives details on the activity of each mercury spike, the specific activity of the spike and the nominal concentration of the radioisotope immediately after the spike (calculated using the volume estimates from Table 1). From the specific activity of the spike, the nominal amount of total mercury added with the spike and its initial concentration can also be determined. The spike addition was made directly into the prop wash of an electric trolling motor to insure rapid dispersal.

FIELD OBSERVATIONS AND COLLECTIONS

The limnocorrals and the bay in which they were situated were monitored weekly or biweekly. Temperature and conductivity profiles were made with a Montedoro-Whitney CTU-3B thermistor and conductivity cell in 1981 and 1982. In 1983 and 1984 these measurements were made with a Hydrolab 4041 Water Quality Monitor which also was used to measure profiles of dissolved oxygen. Secchi disc readings were made with a 25 cm disc divided into black and white quadrants. Results are in Appendix 1.

Water for radioisotope and chemical analyses was collected from the limnocorrals and the bay directly into a weighted, opaque two liter polypropylene bottle which integrated between the water surface and 2.5 m or to a depth 20 cm above the bottom in shallower limnocorrals. The bottle was fitted with a two hole stopper into which two different lengths of stiff polyethylene tubing, one extending to the bottom of the bottle and the other to the end of the stopper, were inserted. The difference in length above the stopper between the intake (shorter) and air outlet (longer) controlled the effective head pressure and rate of filling. The sample bottles were placed in an insulated cooler for transport to the South Indian Lake research station for processing. To collect zooplankton for radioisotope analysis, a 25 L Schindler-Patalas plankton trap with a 85 µm mesh net was used at 0.5, 1.5, and 2.5 m depths and the samples were pooled into a glass jar. The samples were stored on ice until further processing at the South Indian Lake research station.

RADIOISOTOPE COUNTING

Water

In 1981 water samples were routinely fractionated to examine speciation of the radioisotopes (Hesslein et al. 1980; Rudd et al. 1980), and all samples were counted at the Freshwater Institute on a lithium drifted germanium (GeLi) crystal detector mated to a 4000 channel spectral analyzer. Hesslein et al. (1980) describe the system in detail. The water samples were fractionated by passing 500 mL sequentially through a 200 µm prefilter (not retained), a pre-weighed Whatman GFC filter (which was frozen until counted), a pre-weighed 0.45 µm cellulose acetate membrane filter (frozen until counted), a column of cation resin (Dowex AG 50w-x12 100-200 mesh). The eluant was then digested with sulfuric acid and potassium persulfate (Hesslein et al. 1980) and any residual mercury precipitated as HgS. The charcoal, the sulfide precipitate, and the resins were subsequently collected on filter paper and stored in small plastic petri dishes until counted. The results are given in Appendix 2.

In 1982, 1983 and 1984 when only the radioisotope of mercury was added all counting of water was done at South Indian Lake research station on a Packard 3033 Liquid Scintillation Counter (LSC) with automatic external standardization. Fractionation was simplified in these later years. Only 100 mL of sample water was passed through a 0.45 µm membrane filter. The filter, 10 mL of the filtrate and 10 mL of unfiltered water were each counted in PCS (Amersham-Searle) scintillation fluor. The ^{203}Hg results for 1981 to 1984 are in Tables A2.1 to A2.4. The results for the other isotopes added in 1981 are in Tables A2.6 to A2.13. In 1981 only, the net plankton was filtered onto a pre-weighed glass fiber filter, dried at 60°C and counted on the GeLi gamma detector system. In 1982 and 1983 netplankton was collected as in the previous year, filtered onto pre-weighed 0.45 membrane filters, dried at 60°C, reweighed and then counted on the LSC in PCS. All samples were counted in the gel state. When necessary, distilled water was added to the sample to insure that all samples were counted at the same water to fluor ratio and similar counting efficiencies. Net plankton results are in Appendix 4 (Tables A4.1 to A4.7). In 1983 a small number of adult insects caught in emergence traps were dried, weighed and counted on the GeLi system (Appendix 4, Table A4.8). In all years, water samples were counted for tritium content using the LSC and PCS as a fluor. Tritium results are in Appendix 2 (Tables A2.5-A2.8).

Fish

Fish were weighed, measured for total length and most were filleted for the muscle mass while frozen. Muscle mass (less skin) and carcass (remainder of fish) for each fish were analyzed for ^{203}Hg concentration on the GeLi gamma detector. In 1981 only, fish sampled after two weeks and four weeks were analyzed as whole fish less the gut. After ^{203}Hg analysis, muscle samples were analyzed for total mercury. In 1982, six samples from each of two limnocorrals (C82-6 and LM82-3) were dissected for several tissues (muscle, skin, liver, gut, gill and remaining carcass) and the tissues were counted separately on the GeLi detector (Appendix 3, Table A3.31). Stomach contents of these 12 fish were identified and counted. On subsamples of the fish collected from the limnocorrals in 1981, methyl mercury extractions (Uthe et al. 1972) were analyzed to determine the proportion of ^{203}Hg in muscle that was in the methyl form. These samples were counted on a liquid scintillation counter at the Freshwater Institute. Efficiency and decay corrected counts were then compared to dpm/g in the same fish's whole muscle as assayed by the GeLi gamma detector and spectrometer.

Periphyton and sediment

In 1981 four strips (2 cm wide extending to a depth of 2.5 m) of limnocorral wall material were hung at the four compass points within each limnocorral. On August 26 these strips were carefully removed into buckets of lake water and transported to the research station. Ten centimeter lengths were removed from the following depth intervals: 10 cm above the water line to the water line, from the water line to 10 cm below the water line, 50-60 cm, 90-100 cm, 130-140 cm and 170-180 cm. Two strips from the same depth interval but opposite sides of the limnocorral were placed in a plastic petri dish and frozen until counted at the Freshwater Institute on the GeLi detector. Results are in Appendix 4 (Tables A4.9 and A4.10).

In 1981 and 1982 7.5 cm diameter sediment cores were taken with a 3 m plexiglass tube. These cores were sectioned at one centimeter intervals to a depth of 15 cm. The sections were stored in polyethylene bags at 4°C. The sediment sections were transferred quantitatively to aluminum weighing dishes, dried and weighed to determine water content. The dried sediment was then placed in a plastic petri dish for counting on the GeLi detector. Results are in Appendix 4 (Tables A4.11 and A4.12).

TOTAL MERCURY ANALYSES

Total mercury in fish muscle was determined by the atomic absorption spectrophotometry method of Armstrong and Uthe (1971). Water samples for total mercury analysis were taken with a van Dorn sampler and transported to the Freshwater Institute in 300 mL reagent bottles. Sample preservation, extraction and analysis followed closely the procedure of Kopp et al. (1972).

WATER CHEMISTRY

In 1981, ammonia, nitrate, nitrite, soluble reactive phosphate, total dissolved oxygen, oxygen demand, dissolved inorganic carbon, conductivity and pH were done at the South Indian Lake research station using the methods of Stainton et al. (1977). A series of absorbance readings were also made on untreated, filtered (Whatman GFC), and centrifuged (13,000 G) water on a Bausch and Lomb Spectronic 70 spectrophotometer using a 10 cm light path cell at 543 nm and at 455 nm (centrifuged only). All other chemical analyses in Appendix 5 were done by the Analytical Unit of the Freshwater Institute in Winnipeg using the methods of Stainton et al. (1977). In subsequent years, only the oxygen, dissolved inorganic carbon, conductivity, absorbance and pH measurements were made at the South Indian Lake station while all other reported analyses were done at the Freshwater Institute.

RESULTS AND DISCUSSION

All data pertinent to the limnocorral experiments are given in the appendices to this report. Major results of each year's experiments which influenced the evolution of our understanding of the mercury problem will be emphasized in this section. As each year of experimentation built on the previous years results, the experiments will be presented in chronological order.

1981

The 1981 experiments were designed to examine the effects on mercury accumulation in fish of the major water quality changes that had taken place due to the impoundment of Southern Indian Lake (SIL) and Notigi Reservoir. Extensive flooding of the organic soils common in northern Manitoba occurred in both reservoirs. The predominant ground cover is Sphagnum spp. and feather mosses (primarily Hylocomium spp.) and the decomposition products of this moss community, i.e. peat. A moss-peat treatment for limnocorrals was designed based on the ratio of flooded moss-peat mass to reservoir volume of Notigi Reservoir (Newbury et al. 1984). A major water quality change which occurred in SIL, but not in Notigi, was a substantial increase in total suspended solids which originated from the erosion of fine-grained shorelines (Hecky and McCullough 1984). Two limnocorrals were treated with suspended sediment additions at two different rates, termed high (HS) and low (LS). The remaining treatment in 1981 was a nutrient addition which was intended to double the pre-impoundment phosphorus loading rate of SIL which was $0.35\ g\ P\ m^{-2}\cdot y^{-1}$ (Hecky 1975) in order to boost algal production to levels observed in Notigi Reservoir (Bodaly et al. 1984b). Three control limnocorrals (i.e. received no treatment other than the radioisotope spike) were established, two with sewn in plastic bottoms and one with the natural sediment (silty-clay) bottom which was the appropriate control for the moss-peat addition.

There was no significant difference among limnocorrals in growth of yellow perch or emerald shiners (Table 4). However, mortality from unknown causes, i.e. non-sampling, was fairly high, ranging from 50-82% over the course of the experiment (Table 3). The lowest mortality for yellow perch, 59%, occurred in the moss-peat treatment while the highest, 82%, for both yellow perch and emerald shiners, occurred in the nutrient addition treatment. Consequently, the numbers of fish available for analysis at the end of the experiments were limited. However, introduced fish, either yellow perch or emerald shiners or both, were recovered from all limnocorrals at the termination of the experiment. In some cases substantial numbers of extraneous fish were also recovered after the rotenone treatment (Table 4) and these fish were also analyzed for their ^{203}Hg content. The analyses for ^{203}Hg and total Hg for all stocked fish are given in Appendix 3.

The moss-peat treatment (LM81-2) gave the highest accumulation of ^{203}Hg in whole yellow perch and yellow perch muscle while the natural sediment bottom gave the lowest (Fig. 1). Although yellow perch did not survive until the end of the experiment in all limnocorrals, emerald shiners which were only analysed at the end of the experiment were available from all the limnocorrals. The shiners had essentially similar activities of ^{203}Hg as the perch in limnocorrals in which both species survived (Table 7). Using the data for both the perch and shiners together, it is clear that the moss-peat treatment increased accumulation of ^{203}Hg in fish nearly 40x relative to its natural sediment control (C81-1). The fish in plastic bottom controls had ^{203}Hg accumulations in muscle similar to or higher than the natural sediment control. Among the treatments in plastic bottom limnocorrals, fish receiving the nutrient addition and high sediment treatments were similar to their controls while the low sediment treatment was nearly double the highest of the two controls. Clearly the sediment treatments did not give a simple linear response with sediment concentration. The similarity in the perch and shiner results extended also to the extraneous fish caught in the limnocorrals (Table 7) indicating that the results of the experiment were not species dependent.

It is generally found that a high proportion of the Hg found in fish muscle is methyl mercury as this compound is tightly bound to sufhydryl groups in proteins. This assumption was tested for fish sampled from limnocorrals by extracting methyl mercury from muscle tissue which had been previously assayed for its ^{203}Hg content. The extract was then analyzed for ^{203}Hg content and compared with the total ^{203}Hg. These comparisons were made on the last two sampling dates for yellow perch and on the last sampling date for the shiners. Methyl mercury accounted for essentially all the mercury in the muscle of these species at the end of the experiment except in the two plastic bottom controls (Table 8). There was an indication that the proportion of methyl ^{203}Hg to total ^{203}Hg in muscle increased over time based on the two dates available for yellow perch, but the apparent change over time may be within the variance of the analyses. It can be conclusively stated however, that the difference between the ^{203}Hg content of fish muscle in the moss-peat treatment and the other treatments is due to methyl mercury.

The increases in ^{203}Hg content in fish over time which all the limnocorrals exhibited do not necessarily mean that total mercury concentrations have similarly increased. The ^{203}Hg isotope of mercury does not occur naturally so the ^{203}Hg content of all the fish at the start of the experiment is zero, and the content of ^{203}Hg must increase over time as the mercury within the fish approaches equilibrium with the ^{203}Hg source. Because of the small numbers of yellow perch left in the limnocorrals and the consumptive requirement for the methyl mercury analyses discussed above, the final total mercury, i.e. radioactive and cold mercury, can only be assessed for two limnocorrals in 1981, the moss-peat treatment and one of the plastic bottomed controls. Muscle mercury content for these fish are compared to the time-zero muscle mercury concentrations in Table 9. Fish from the moss-peat limnocorrals had their muscle mercury concentration increase by 5x over the course of the experiment while the mercury in fish from the control limnocorral declined. In both cases the specific activities (dpm ^{203}Hg per mg total Hg) of fish muscle were not substantially different between the last two sampling dates within each limnocorral indicating that equilibrium between ^{203}Hg and natural mercury may have been reached. If equilibrium was reached, then substantial amounts of natural mercury were available to the fish in both cases as the specific activity of the fish is well below that of the radinisotope spike (Tables 5 and 9) indicating dilution by "cold" (non-radioactive) mercury in each case. However, the data from 1981 alone are too fragmentary to make a firm conclusion about the relative availability of natural mercury. For example, given the evidence that there was no change in the total mercury in fish muscle in the control, it is as plausible that ^{203}Hg labelling of the muscle mercury pool was extremely slow, so slow that the differences in the last two sampling periods were statistically indistinguishable. If this were the case, then no conclusion about natural mercury availability can be reached based on the 1981 data alone.

In 1981 a great deal of effort was expended examining the speciation of ^{203}Hg within the water column. For example the activity in the dissolved form was operationally split into cation, anion, organic and undetermined forms (Appendix 2). Also the fractionation of particulate ^{203}Hg was similarly operationally defined by particle size (Appendix 2). The moss-peat treated limnocorral was not unusual in any of these aspects compared to the other limnocorrals. It must be recognized that none measure methyl mercury which may be only a small of these operational parameters specifically measure methyl mercury which may be only a small component of the ^{203}Hg in any of the fractions. As this is the only mercury compound which accumulates in fish muscle, the lack of simple correlation with total mercury in these different fractions is not surprising. Consequently, the fractionation analyses were largely discontinued in subsequent years.

1982

The 1982 experiments had three major hypotheses: 1) the moss-peat effect was dose-dependent, i.e. more moss-peat would yield higher mercury uptake, 2) the effect of moss-peat material would attenuate with time, and 3) high suspended sediment concentrations would increase mercury uptake. The dose effect was investigated by adding moss-peat material at two different rates, the LM treatment of 1981 and double the LM treatment (high moss-peat, HM). The effect of time on the moss-peat effect was assessed by using the 1981 moss-peat material and limnocorral again and by installing a limnocorral over a bed of moss-peat material originally flooded in 1976. The effect of suspended sediments was tested in a limnocorral with a natural sediment bottom (cf. plastic bottoms in 1981) at a rate of suspended sediment addition comparable to the high rate of 1981. Details of the 1982 treatments are given in Table 2.

The moss-peat treatments did give an approximately proportional response in fish muscle mercury concentration. At the end of the experiment the mean total mercury concentration in fish in the LM treatment was 0.026 µg·g^{-1} higher than the controls while the HM treatment was 0.053 µg·g^{-1} higher, i.e. a doubling in response (Fig. 3). The difference in the ^{203}Hg response was less dramatic (Fig. 4) indicating a greater dilution of the radioisotope spike by natural mercury in the HM treatment. Comparing the specific activities of the LM and HM to the specific activity of the radioisotope spike indicates that at the end of the experiment about 1.7x as much natural mercury had been incorporated into fish muscle in the HM treatment as in the LM treatment. This indicates that the availability of natural mercury was also nearly proportional to the mass of the moss treatment. However, because the specific activities of the mercury in fish muscle had not obviously reached a plateau value which would indicate equilibration with available ^{203}Hg (Fig. 5), no definitive statement can be made about the exact proportion of natural mercury mobilized in the treatments.

The results of the time course experiment indicated that after one year of immersion (the moss-peat originally added in 1981) the moss-peat material still had a positive effect on bioaccumulation of mercury in fish muscle. This one year old material mobilized substantially more total Hg than any other treatment in 1982 (Fig. 3). However, the mobilization of ^{203}Hg was less than the new moss-peat treatments (Fig. 4) indicating that most of the mercury mobilized did not arise from the mercury added in the 1982 radioisotope spike. Possible sources of this non-spike mercury were the degradation of the moss-peat material, natural sediment mercury and/or residual mercury added in the 1981 spike. The most appropriate control for this limnocorral was OC82-5 which had also served as a control in 1981. This limnocorral, like OM82-8, received a radioisotope spike in 1981 as well as in 1982. However, OC82-5 was indistinguishable from the newly established control C82-6 in bioaccumulation of ^{203}Hg or total Hg, and this suggests that the one year old moss-peat was significant as a source of mercury as well as stimulating methylation of the available mercury. The specific activity of the mercury in fish muscle in OC82-5 apparently plateaued or possibly began to decline by the end of the experiment (Fig. 5) indicating that equilibration with mercury sources was achieved in this case. Comparing the maximum specific activity with that of the spike shows that 95% of the mercury mobilized into fish muscle was not added with the 1982 spike. In contrast with the one year old moss-peat material (OM82-8), the flooded land surface which had been submerged for six years (FM82-1) did not stimulate bioaccumulation of either ^{203}Hg or total Hg. Unfortunately, conclusions about the effect of length of submersion were called into question by the 1983 experiments which indicated that it is difficult to compare results in a new limnocorral with a reused limnocorral.

The high suspended sediment treatment which was added to a limnocorral (HS82-7) with a natural sediment bottom in 1982 increased bioaccumulation of ^{203}Hg and total Hg (Fig. 3 and 4). The total Hg concentration in fish muscle at the end of the experiment was comparable to that of the high moss-peat treatment (HM82-4) although the ^{203}Hg uptake was only about one-half (relative to controls) that of HM82-4. The specific activity of the fish muscle mercury peaked in mid-August (Fig. 5) and at its maximum it was only 16% of the specific activity of the spike. The addition of suspended sediments both stimulated methylation relative to controls and increased the availability of natural mercury in the limnocorral.

1983

The 1983 set of limnocorral experiments was intended to examine the recovery of previously treated limnocorrals and a possible mitigation treatment. In addition, the replicability of a treatment, as opposed to controls, was determined and the effect of another organic material (spruce boughs) was to be investigated. The treatments are given in Table 2.

A number of problems occurred in 1983 which weaken the interpretation of results. Firstly, 1+ yellow perch were not caught in Wupaw Bay. Consequently, stocking densities were much lower than in previous years, and 2+ and 0+ perch were added to the limnocorrals. As the 2+ perch were more comparable to the size of fish used in other years, only the results for those fish are presented here. To insure that there would be fish recovered at the end of the experiment, fish were only sampled three times over the summer. In fact, survivorship was good and adequate samples were recovered from all limnocorrals (with one exception discussed below). The second departure from previous years' experiments was the high proportion of limnocorrals which were reused, i.e originally used in 1982 or 1981 in the case of the old moss-peat treatment. These reused limnocorrals had thereby received at least two spikes of mercury attendant with radioisotope addition (Table 2). Methylation/demethylation assays performed in the limnocorrals in 1982 and 1983 (Ramlal et al. 1987) indicated that the microbial communities of these reused limnocorrals differed from new limnocorrals in their predilection towards net methylation of mercury. This makes it impossible to compare new and reused limnocorrals because multiple radioisotope spikes eventually create mercury contaminated systems. The third problem was that the spruce treatment went anoxic and all the fish were killed; therefore no conclusion could be reached concerning the effect of this treatment on mercury net methylation and accumulation in fish.

The 1983 experiments did establish that the variance in treatment limnocorrals as well as controls, which had been run in pairs in previous years, was low (Fig. 6 and 7). The mean ^{203}Hg activity in fish at the end of the experiments in limnocorral LM83-3 was 124,500 dpm$\cdot$g^{-1} while it was 125,400 dpm$\cdot$g^{-1} in LM83-7; there was more variance in total mercury, 0.16 µg$\cdot$g^{-1} compared to 0.13 µg$\cdot$g^{-1}.

The two new moss treatments can be considered as controls for the sediment-moss treatment LMS83-8. This treatment was meant to explore the mitigatory effect of resuspending lake (as opposed to bank) sediment on mercury bioaccumulation. Rudd and Turner (1983) concluded that sediment suspension and subsquent deposition could ameliorate the fish mercury problem in Clay Lake, Ontario. As our 1982 experiments had already demonstrated that suspension and deposition of bank sediment enhanced mercury bioaccumulation, we decided to use offshore sediments from Long Bay to determine if they would have an ameliorative effect over flooded moss. In fact, the suspension and deposition of lake sediments enhanced the bioaccumulation of ^{203}Hg (Fig. 6) and had a negligible effect (or slightly enhanced) on total mercury in fish muscle (Fig. 7). The higher specific activity (Fig. 8) of the Hg in fish muscle in LMS83-8 as compared to LM83-3 and LM83-7 indicates that the sediment suspension made more of the mercury from the spike available for methylation while it may have reduced the input of natural mercury from the moss-peat material and bottom sediments of the limnocorral. The addition of lake sediments in suspension may have partially shifted the locus of methylation from the bottom materials toward the water column, but it did not reduce the bioaccumulation of mercury over the period of the experiment.

1984

The 1984 experiments compared effects of three different organic substrates, moss-peat, spruce boughs (at a reduced dose from 1983) and prairie sod. Also the effect of spike mercury concentration was examined by tripling the amount of mercury added with the radioisotope spike. In addition, two limnocorrals,one receiving a low moss-peat dose and one control did not receive any radioisotope spike or added mercury in any form. This pair of limnocorrals was to determine if the addition of ionic mercury was a causative agent of enhanced bioaccumulation in former experiments. Details of the limnocorral treatments are given in Table 2. The experiments started in the first week of July as in all other years, but fish sampling was more frequent and extended until the end of September. Fish survivorship was excellent in all limnocorrals, but fish growth was significantly better in the prairie sod treated limncorral (Fig. 9).

In the limnocorrals which received no mercury addition (LMN84-1 and CN84-2), the moss-peat treatment enhanced mercury accumulation in fish by a factor of 1.7 (Fig. 10). In the moss-peat treatment which received a mercury addition (LM84-5), the total mercury concentration in fish muscle at the end of the experiment was higher with an enhancement factor of 2.4 over CN84-2. However, up to the end of August there was little difference between the moss-peat treatments with and without a mercury spike indicating that the added mercury was not required to cause the response in fish muscle mercury, although it may have prolonged the period of uptake.

The addition of extra mercury with the radioisotope spike did not have any effect on total mercury uptake by fish muscle until the last sampling period (Fig. 10). It did reduce the specific activity of the Hg taken up indicating that the extra mercury was available to the methylating bacteria (Fig. 11). In the first season of use, the limnocorral responses appear to be independent of the mercury added with the spike. However, the increased uptake in the limnocorral with the high dose of ionic mercury on the last sampling date suggests that time may be a factor in the effect as Ramlal et al. (1987) concluded when they observed that reused limnocorrals had higher rates of methylation than new ones regardless of treatment.

The three organic substrates used in 1984 all caused greater bioaccumulation of mercury in fish muscle relative to controls (Fig. 10). Total mercury concentration was substantially higher in the LM84-5 and S84-3 than in the prairie sod treatment SD84-4, but this difference may have been caused by the much higher fish growth rates in the sod treatment which would tend to dilute the mercury taken up. Regardless, the conclusion would be that materials of the boreal forest are efficient substrates for mercury bioaccumulation in fish muscle. The sod result suggests that regions with different vegetation may differ in degree from boreal forest reservoirs, but the qualitative effect on fish mercury is likely to be universal, i.e. all flooded soils and vegetation will stimulate mercury methylation by bacteria and enhance bioaccumulation of Hg. The specific activity of the Hg in fish muscle rose in all three treatments and apparently reached a plateau suggesting equilibration between the added ionic mercury and other mercury sources available for methylation and accumulation in fish. The added substrates differed significantly in their effect on the contribution of natural mercury to fish muscle. The sod and spruce treatments mobilized 83-86% of the mercury accumulated in fish muscle from natural sources (only 14-17% of the mercury in fish was derived from the spike) while the moss-peat treatment mobilized 93% of the mercury from natural sources. Twice as much natural mercury was mobilized by the moss-peat treatment compared to sod and spruce. Therefore, although all three substrates caused net methylation and bioaccumulation of mercury, the moss-peat material was more effective in mobilizing natural mercury than the other treatments. In all the controls, the amount of natural mercury available substantially diluted the spike (e.g. Table 9), indicating that much of the mercury was coming from the lake sediments within the limnocorrals. These same sediments underlaid all the limnocorrals. Clearly the primary effect of the added materials was to disturb the methylation/demethylation balance occuring within the limnocorrals. The effect of the materials or the radioisotope spike on the total mercury available for cycling through the methyl mercury pool was minor.

ANNUAL VARIABILITY

Moss-peat limnocorrals were run in every year and new control limnocorrals were run in three years. When these are compared (Fig. 12 and 13), it is clear that mercury uptake by fish in the control limnocorrals was generally consistent from year-to-year especially in terms of radioisotope uptake. In the moss-peat treatment, three of the four years were quite similar in both radioisotope uptake and total Hg uptake into fish. The 1981 moss-peat limnocorral had by far the most dramatic response of all treatments in all years. Efforts were made to run the experiments in a very similar fashion, i.e. similar timing, similar treatments, similar radioisotope spikes in all years, etc. and it seems unlikely that the 1981 extreme can be accounted for by experimental technique. Rather, limnological conditions which did vary from year-to-year in the bay likely played a modifying role in mercury bioaccumulation (Table 10). We did not conduct any experiments to specifically address what factors may be significant, and further research in this direction would be warranted to improve our ability to predict the effect of reservoir conditions on fish mercury concentrations.

ACKNOWLEDGMENTS

Funding for these experiments was provided by the Department of Fisheries and Oceans and the Canada-Manitoba Mercury Agreement. The study design and choices of hypotheses to be tested were strongly influenced through numerous conversations with J.W.M. Rudd, M.A. Turner, K. Patalas, R.H. Hesslein and P. Campbell. R.H. Hesslein provided access to the gamma spectrometer and supervised the counting of material on it. Paul Wilkinson, assisted by Paul Campbell, performed the radioisotope spikes in 1981, 1982 and 1983. We would like to thank A. Lutz and R. Hunt of the Freshwater Institute Heavy Metals Laboratory for performing the mercury analyses. The Analytical Unit of the Freshwater Institute performed all the major ion and particulate analyses as well as many of the nutrient analyses. Aspects of the study were conducted by Agassiz North Consultants under DSS contracts OSF82-00027 and OSF83-0004.

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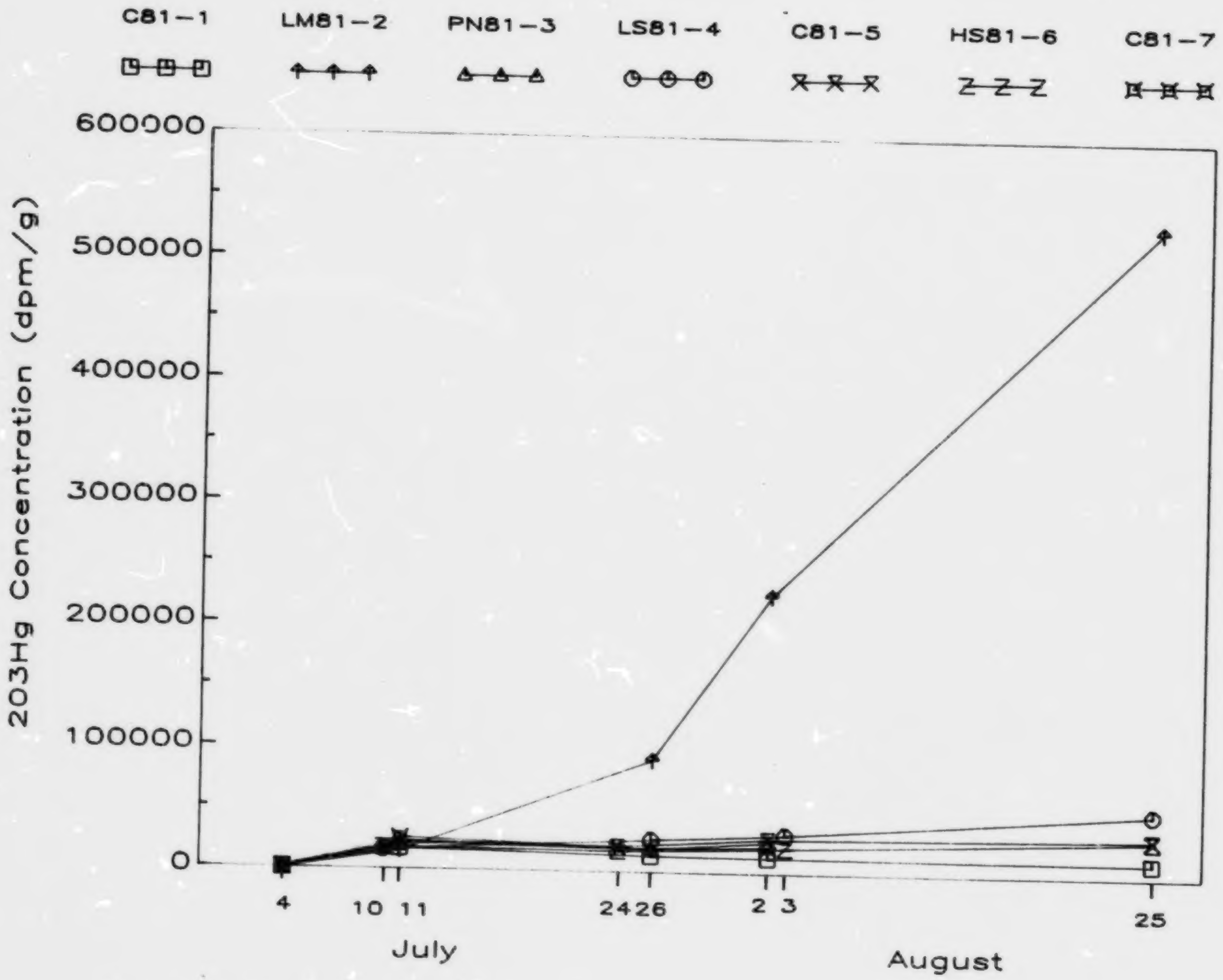

Fig. 1. Concentration of ^{203}Hg in combined yellow perch muscle and carcass, 1981.

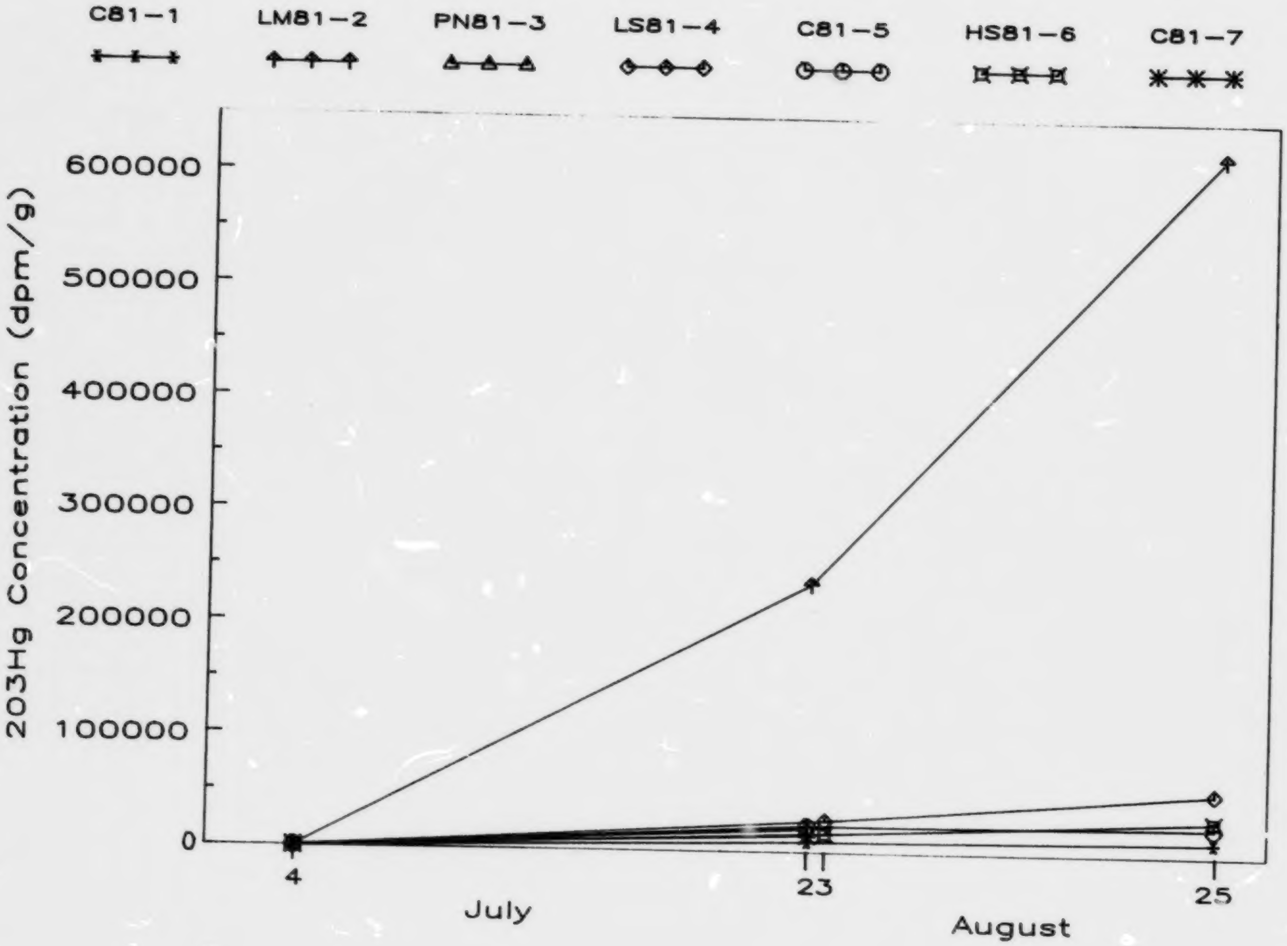

Fig. 2. Concentration of ^{203}Hg in yellow perch muscle, 1981.

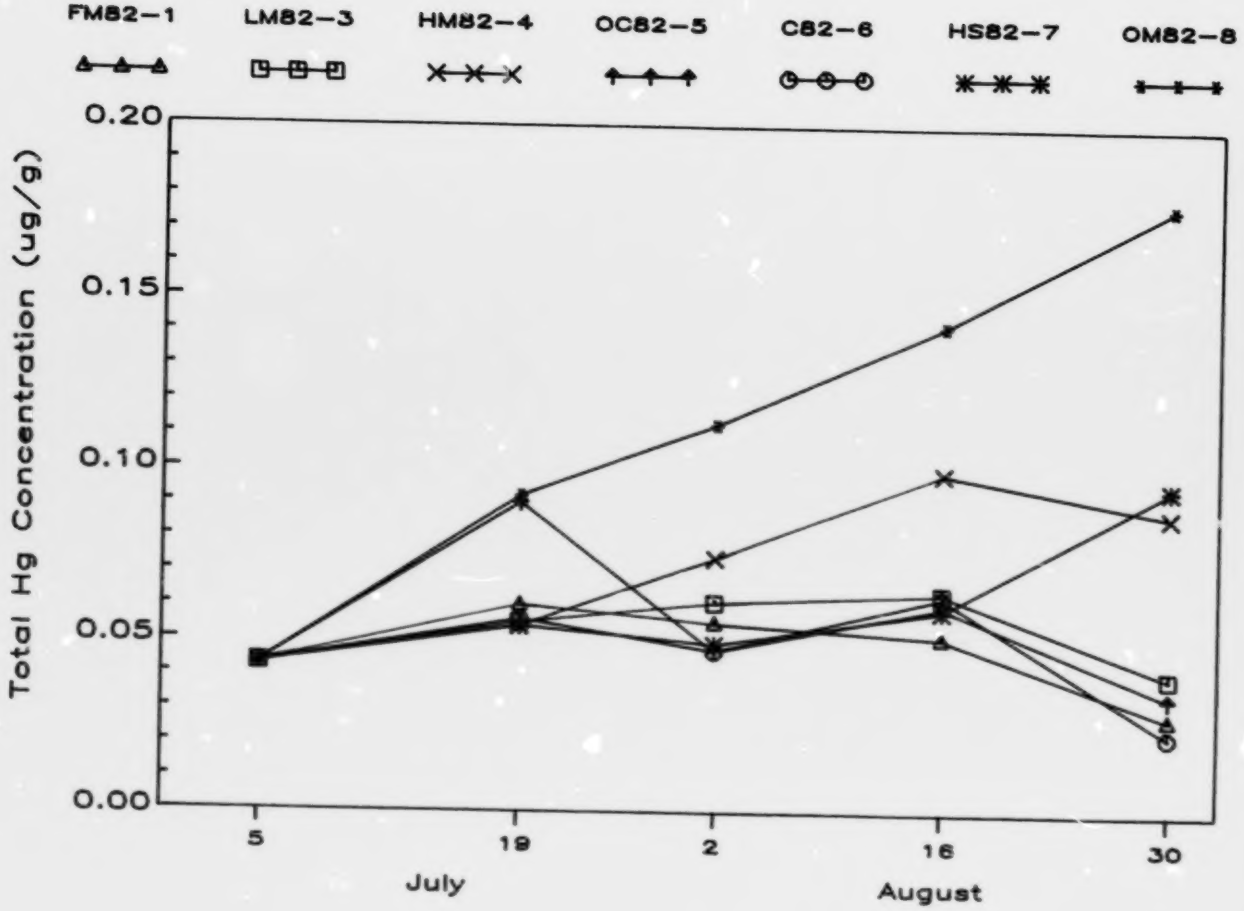

Fig. 3. Concentration of total Hg in yellow perch muscle in 1982.

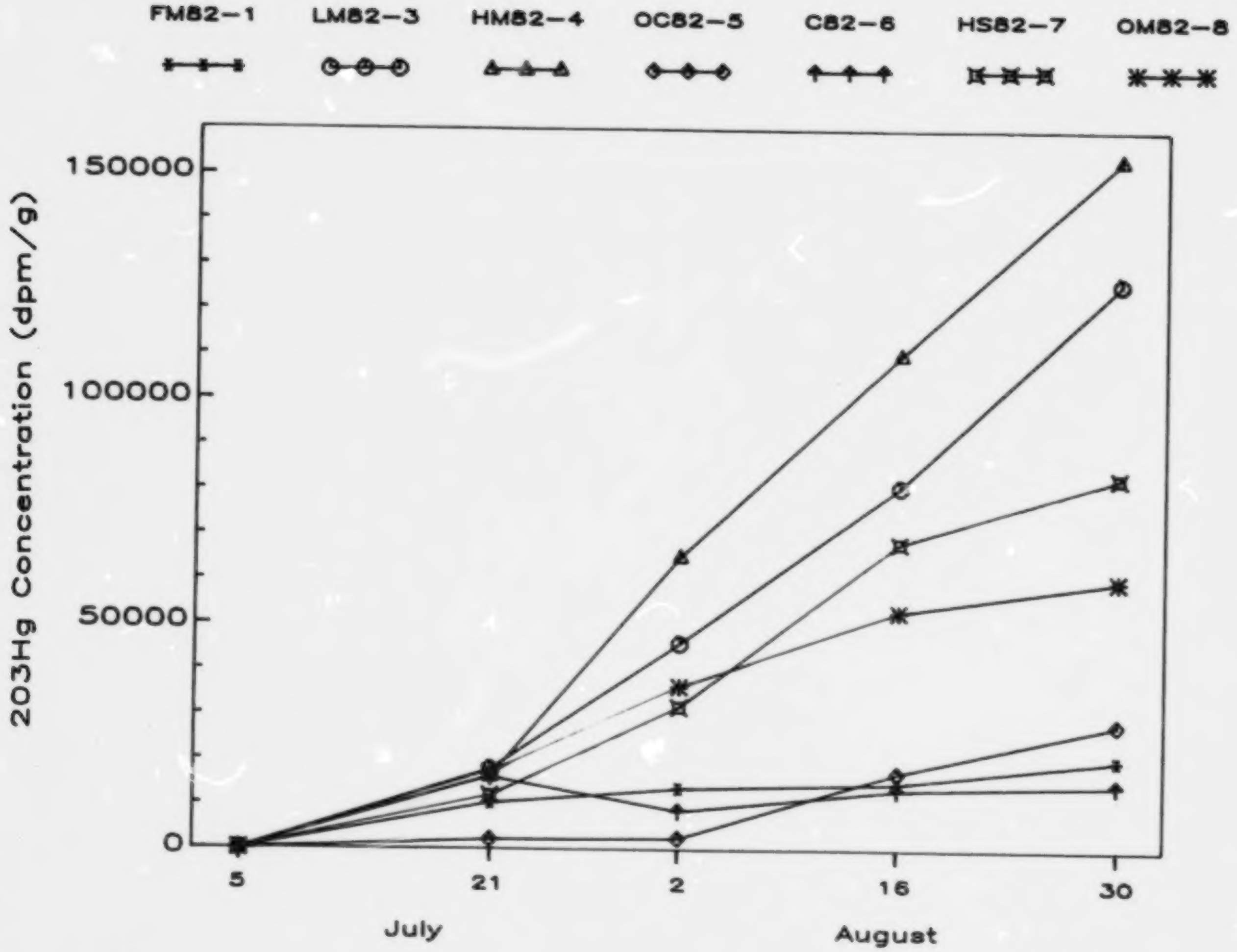

Fig. 4. Concentration of ^{203}Hg in yellow perch muscle in 1982.

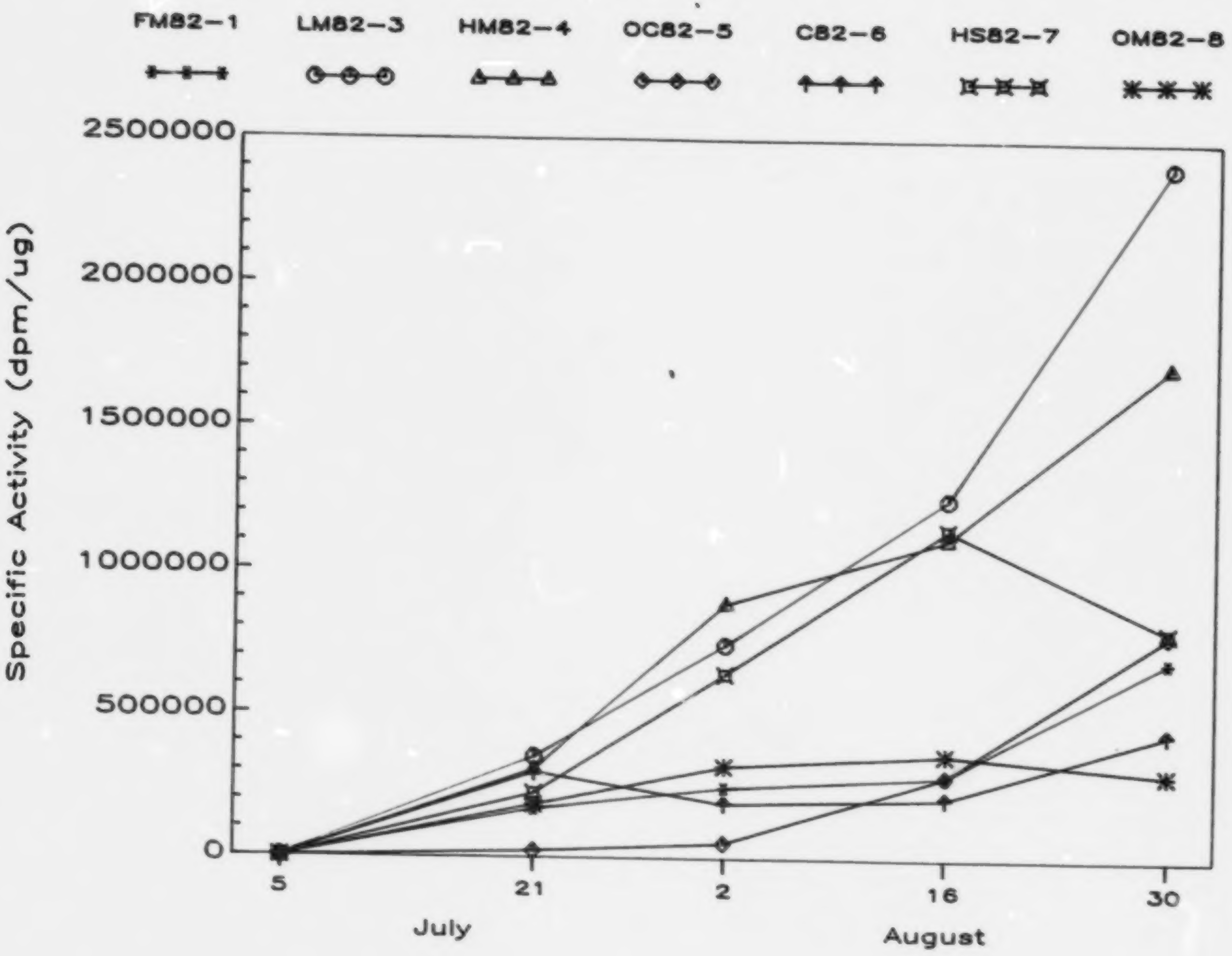

Fig. 5. Specific activity of ^{203}Hg in yellow perch muscle in 1982.

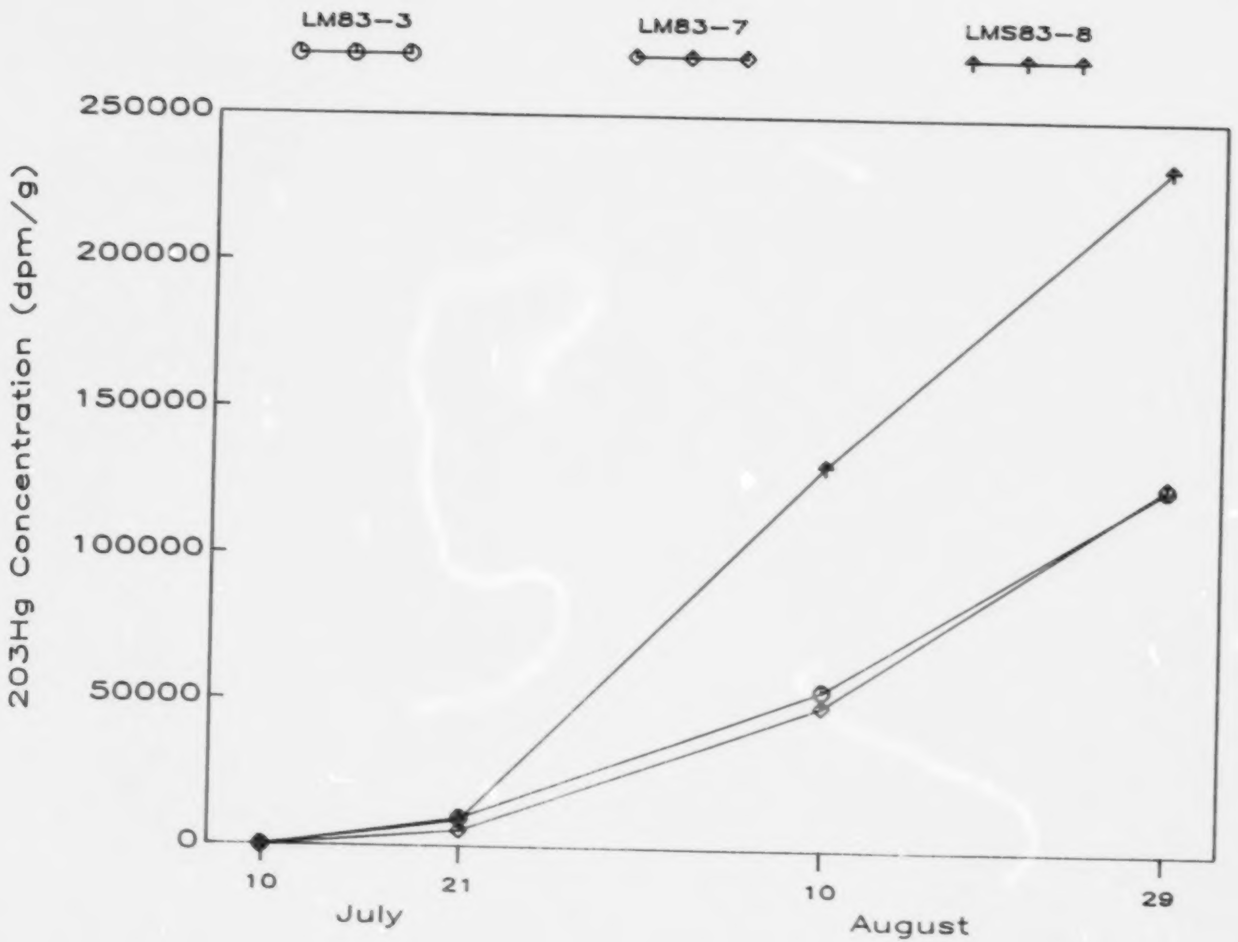

Fig. 6. Concentration of ^{203}Hg in yellow perch muscle in low moss-peat limnocorrals, 1983. Lake sediment was added in suspension with moss to LMS83-8.

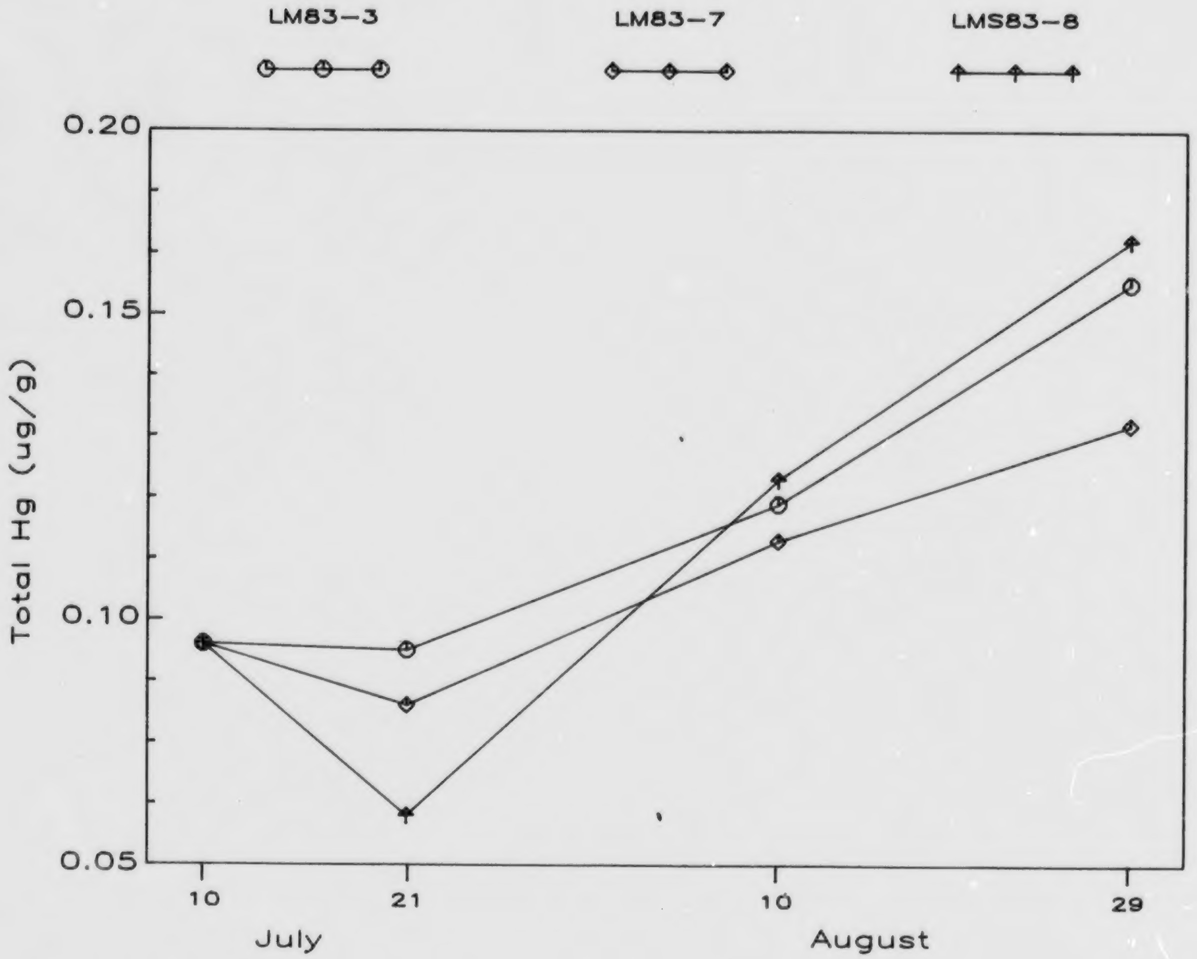

Fig. 7. Concentration of total Hg in yellow perch muscle in low moss-peat limnocorrals in 1983.

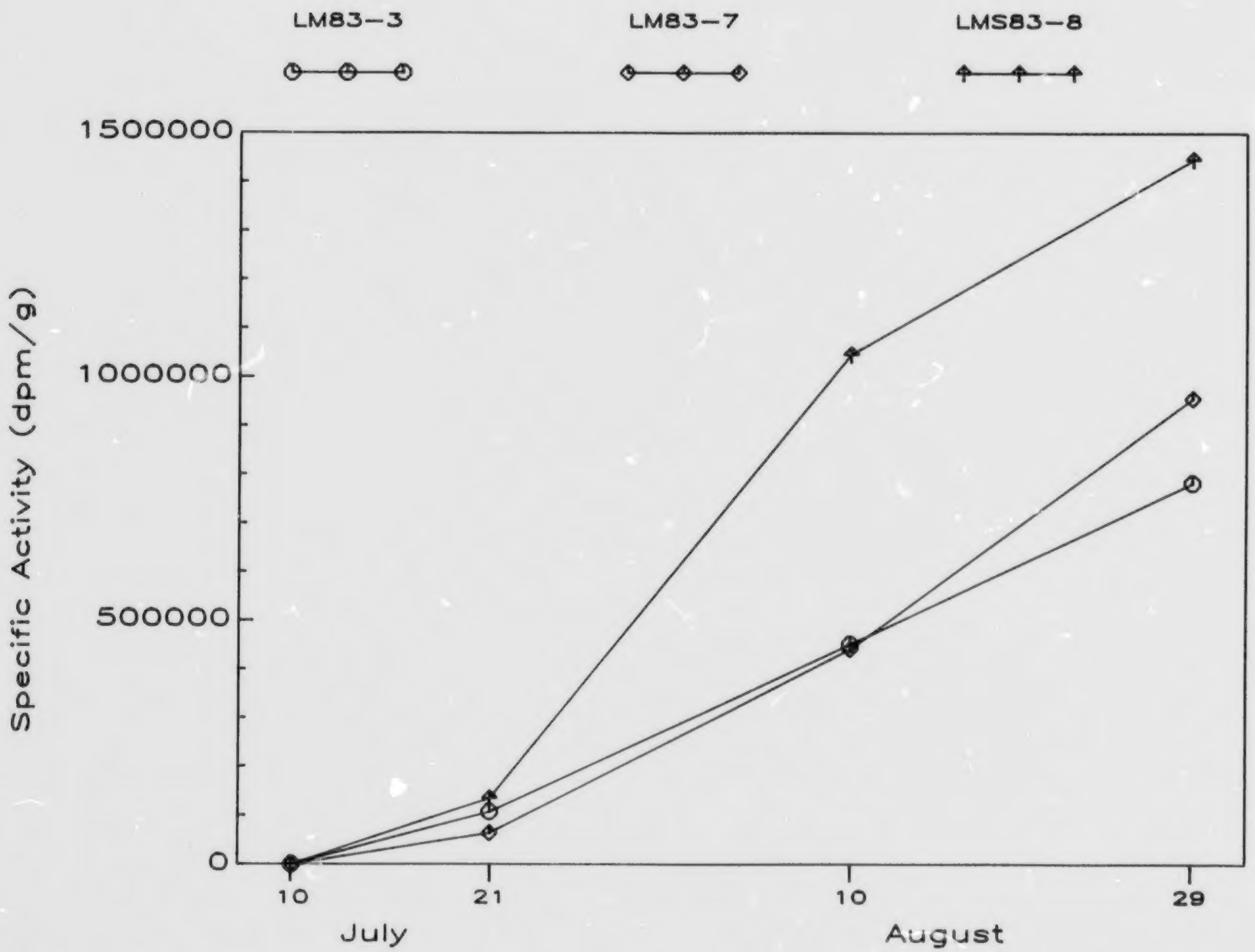

Fig. 8. Specific activity of ^{203}Hg in muscle of yellow perch from low moss-peat limnocorrals, 1983.

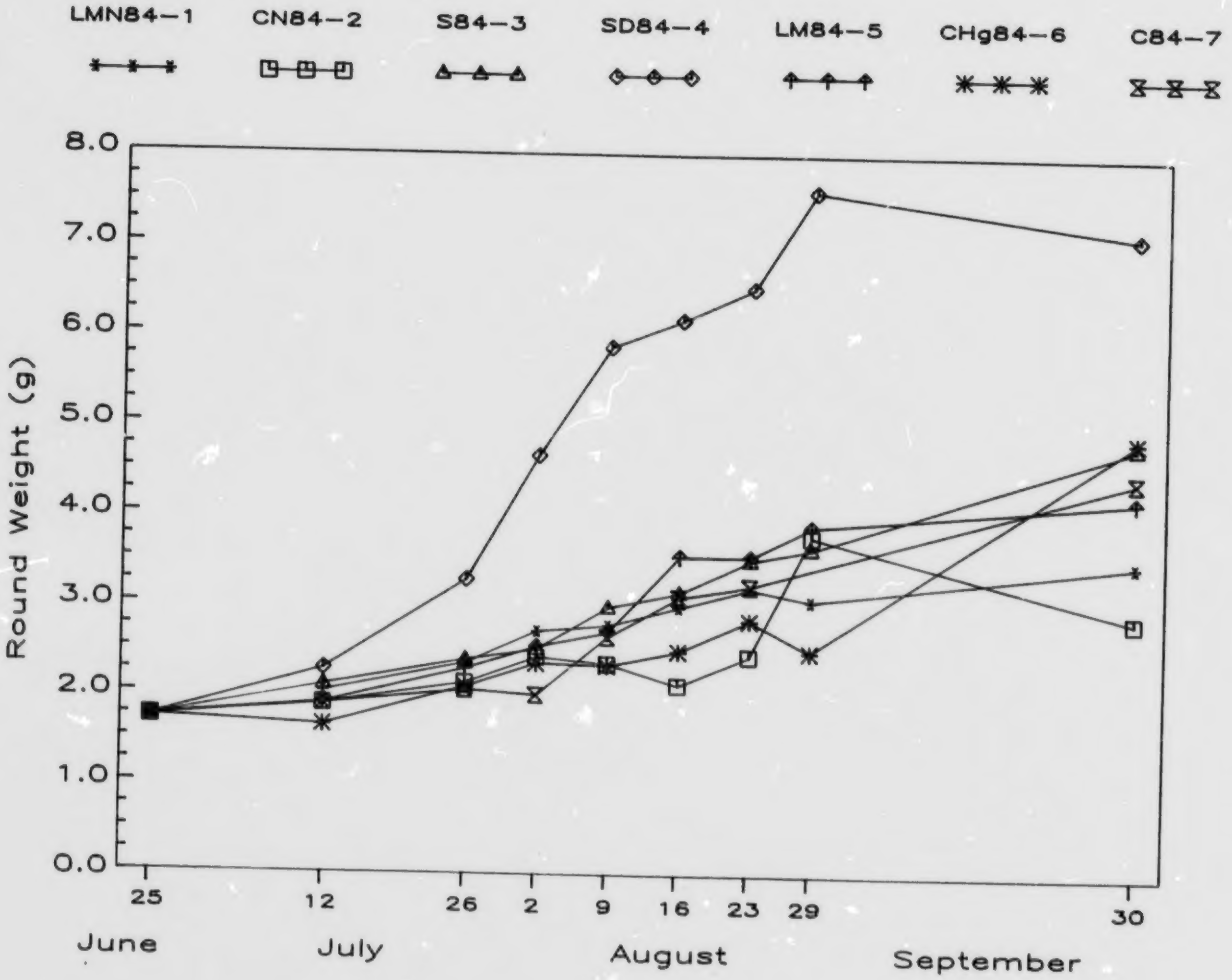

Fig. 9. Fish growth in limnocorrals in 1984.

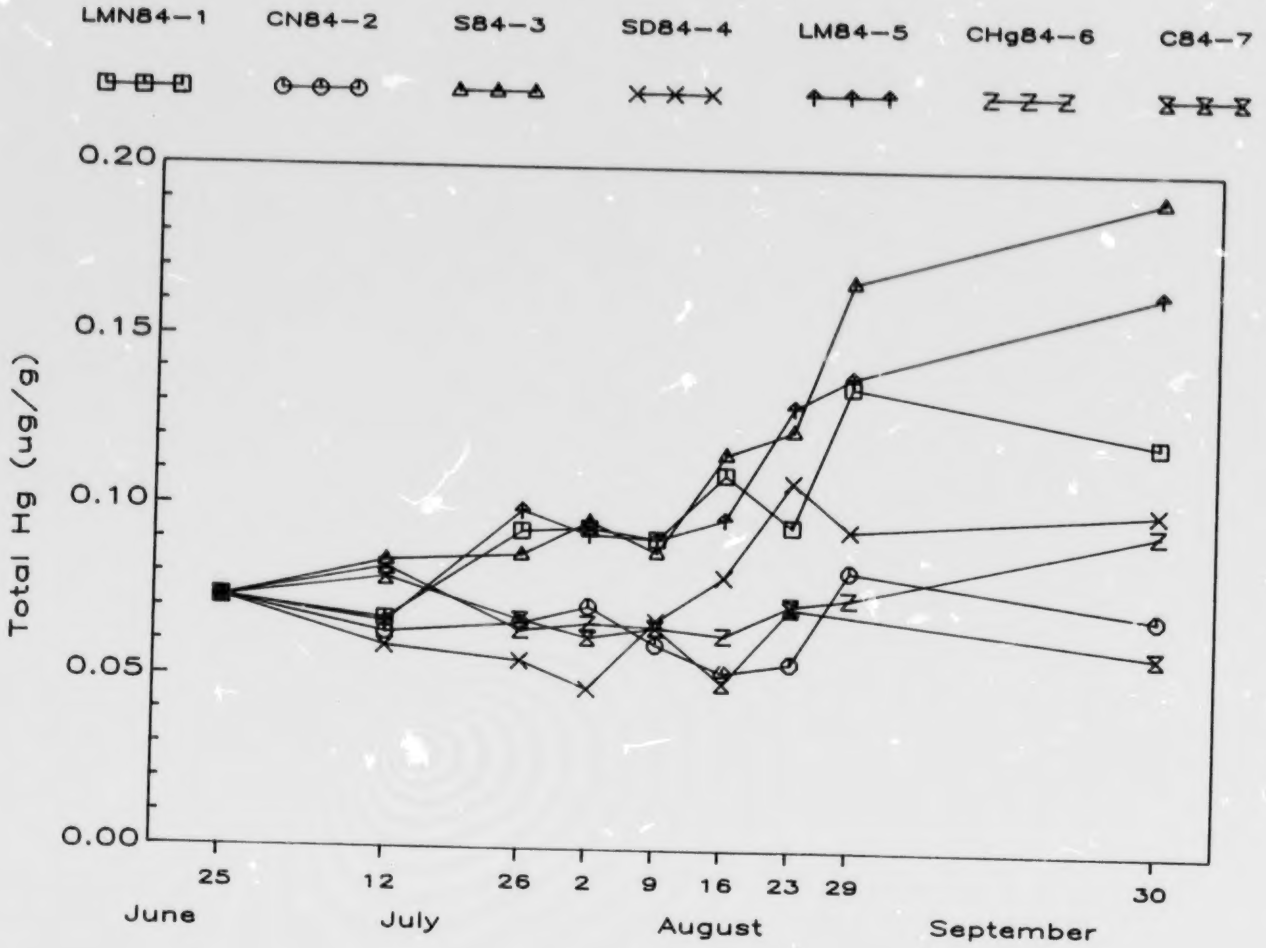

Fig. 10. Concentration of total Hg in yellow perch muscle in 1984.

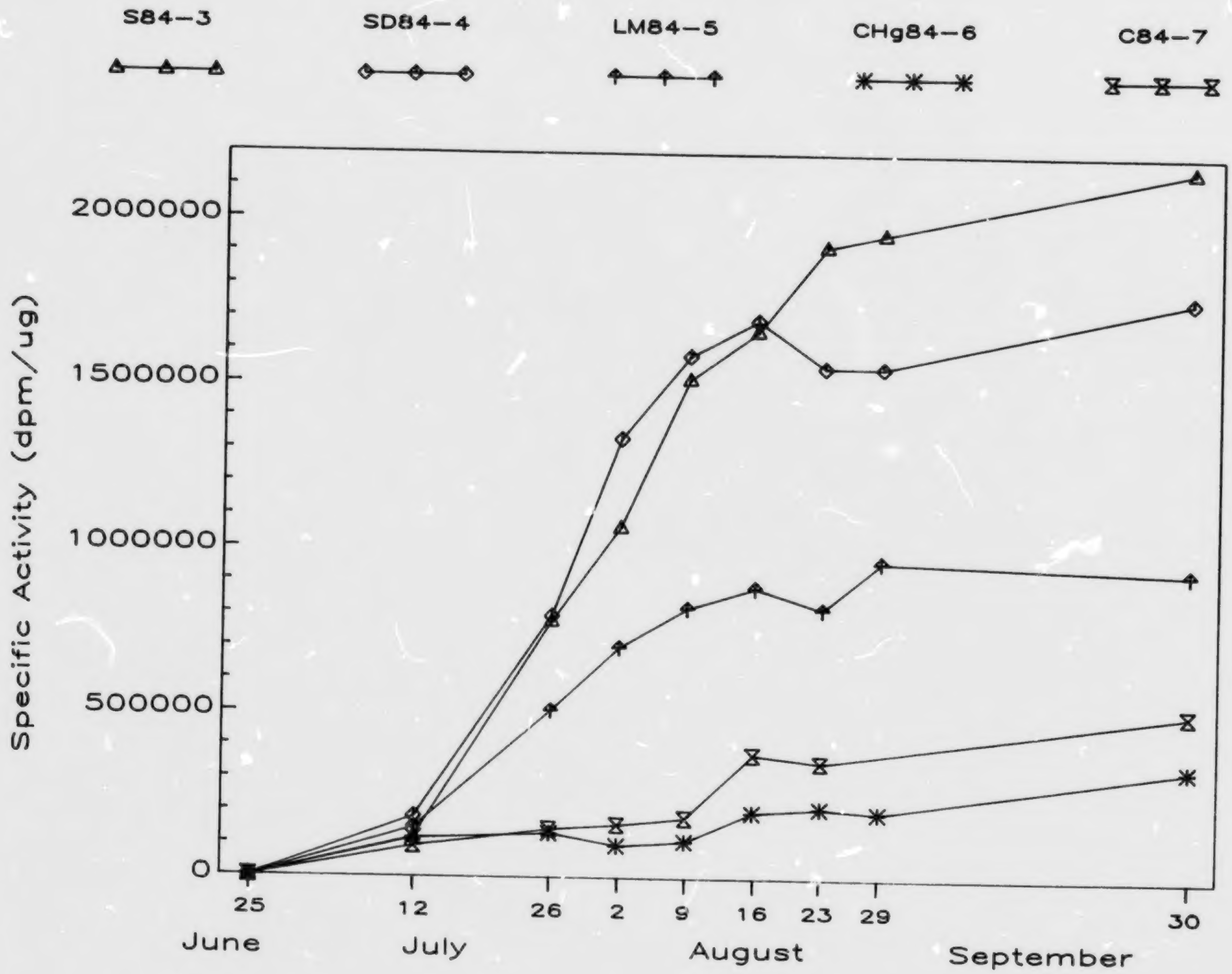

Fig. 11. Specific activity of ^{203}Hg in muscle of yellow perch, 1984.

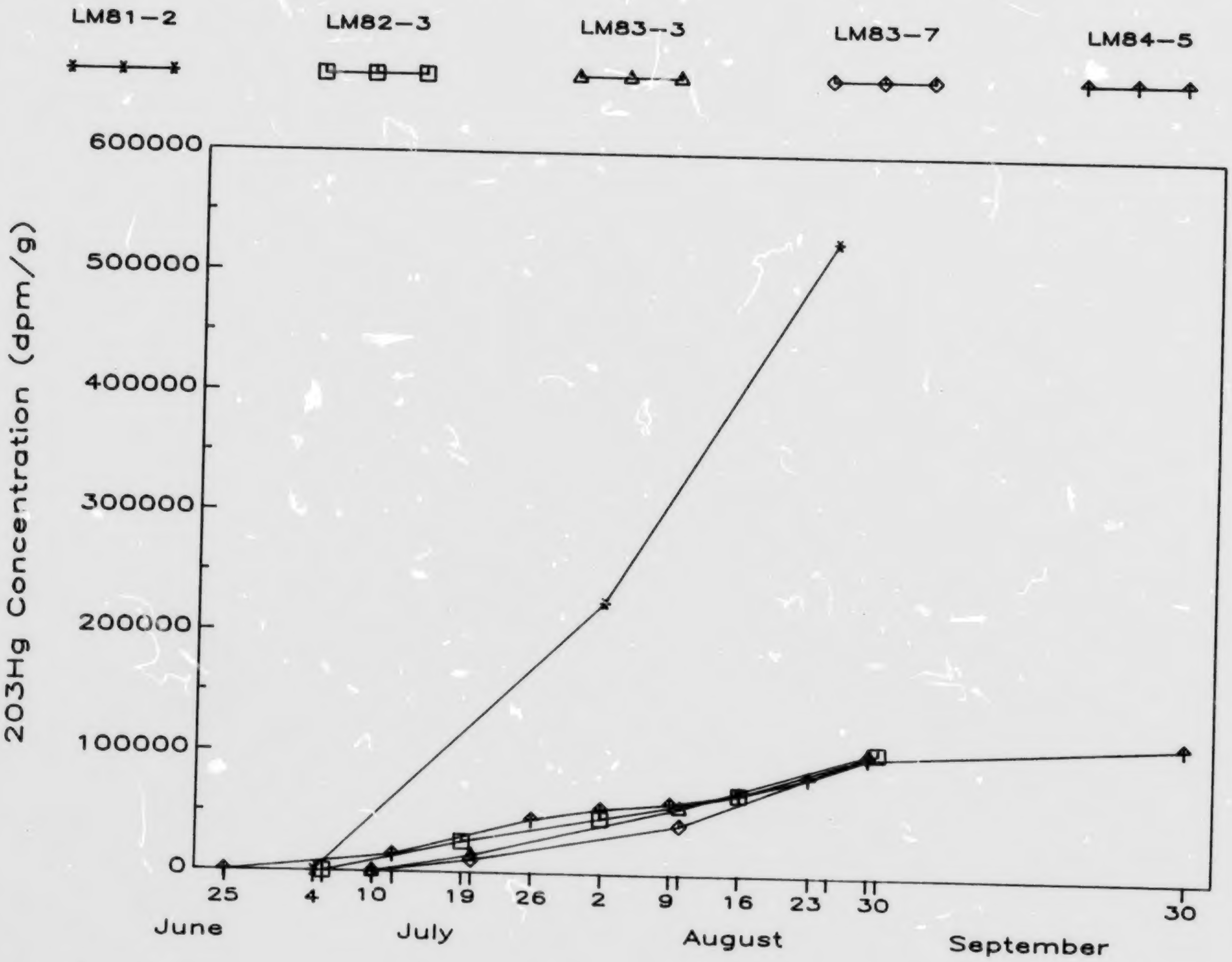

Fig. 12a. Comparison of ^{203}Hg uptake by yellow perch muscle in low moss-peat limnocorrals, 1981-84.

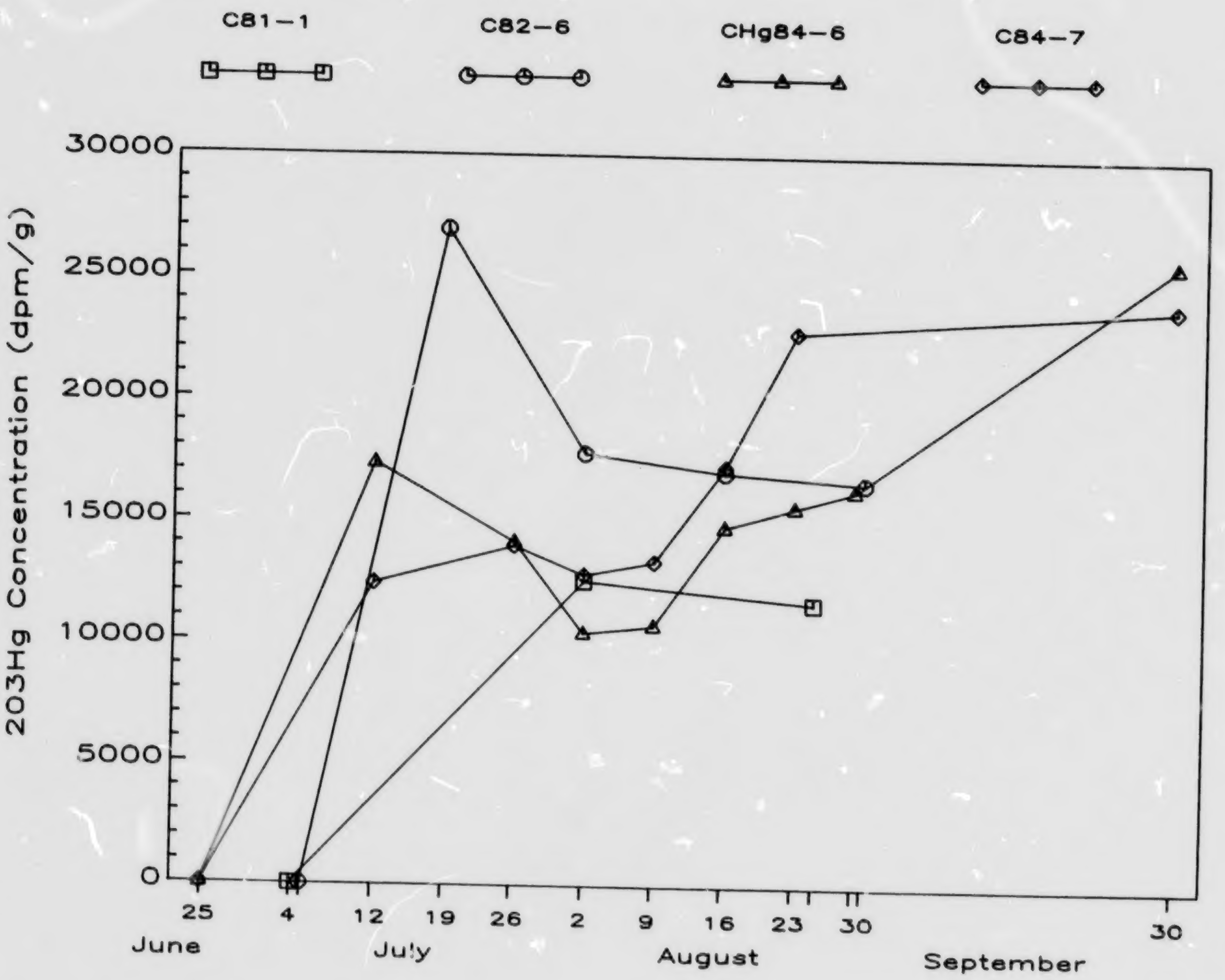

Fig. 12b. Comparison of ^{203}Hg uptake by yellow perch muscle in control limnocorrals, 1981-84.

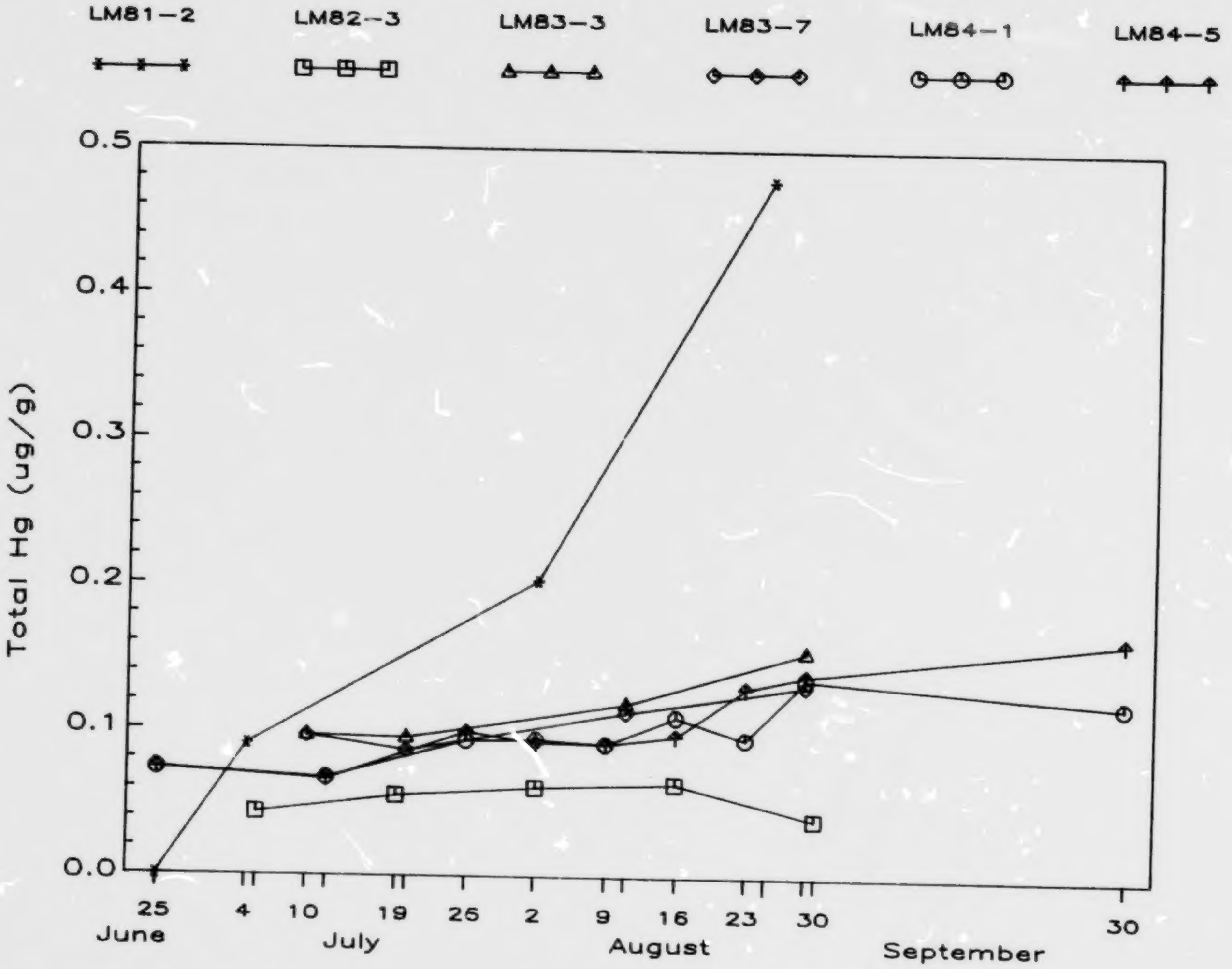

Fig. 13a. Comparison of total Hg uptake by yellow perch muscle in low moss-peat limnocorrals, 1981-84.

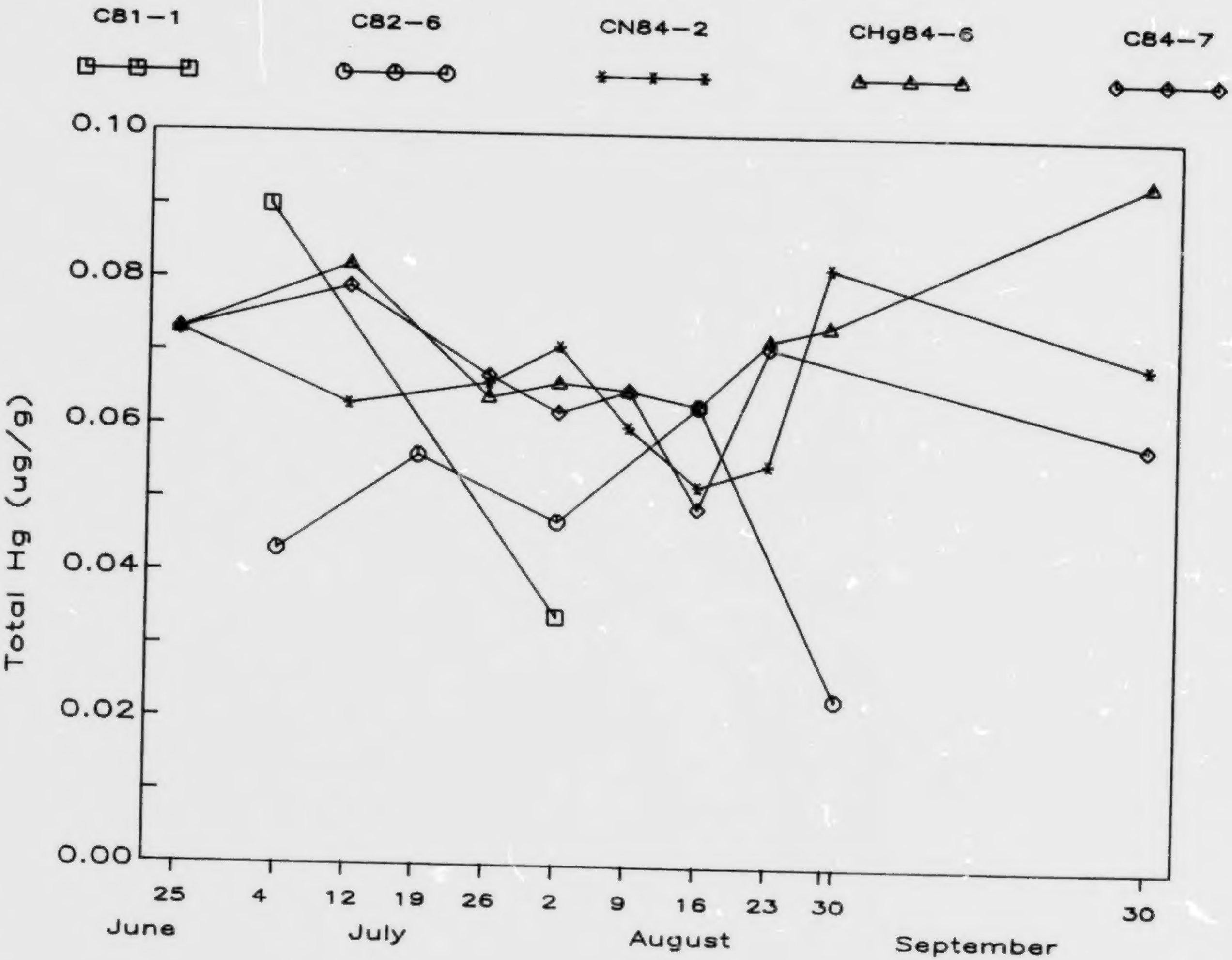

Fig. 13b. Comparison of total Hg uptake by yellow perch muscle in control limnocorrals, 1981-84.

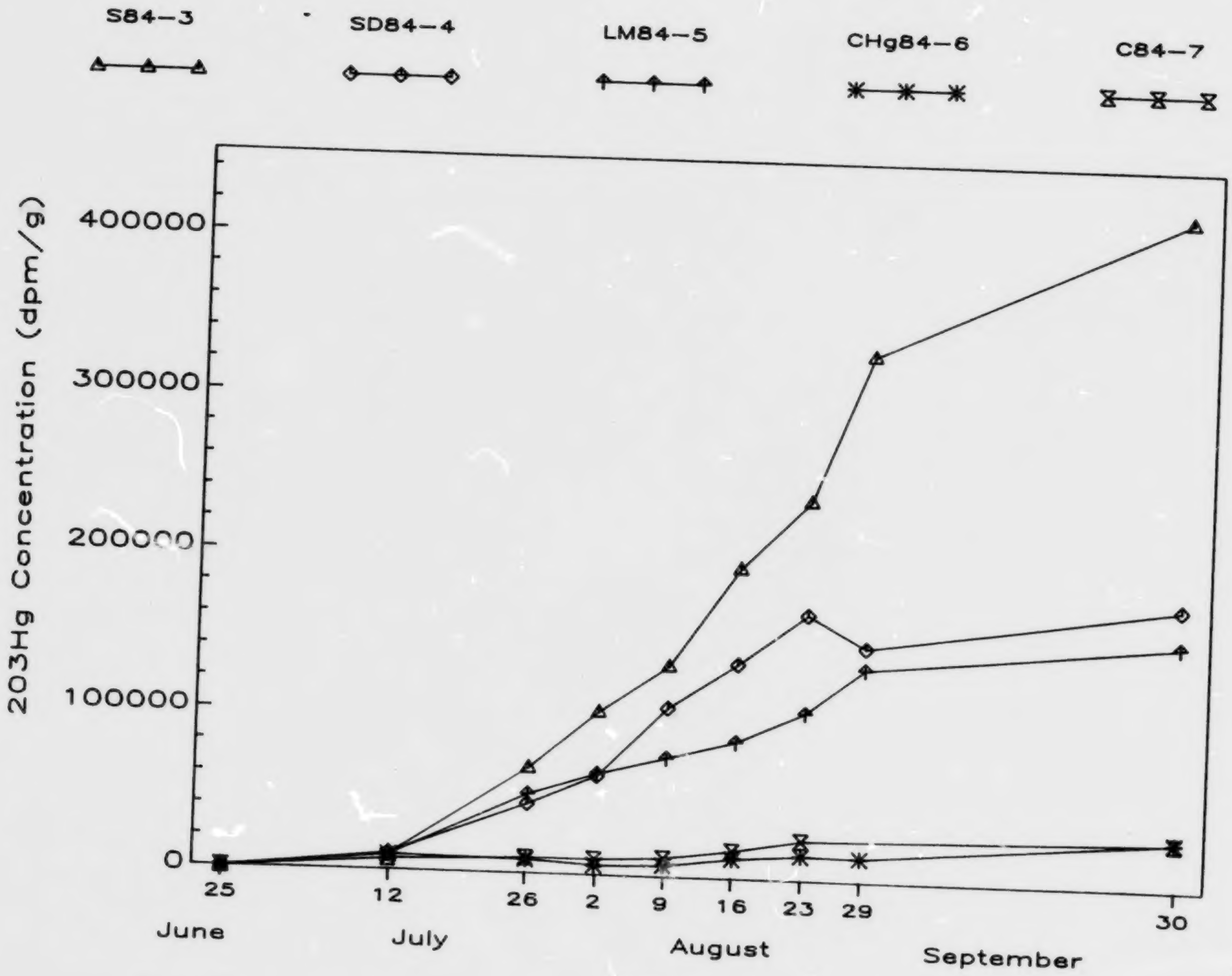

Fig. 14 Concentration of ^{203}Hg in yellow perch muscle in 1984.

Table 1. Limnocorral designators, treatment names, bottom type, volume, and mean depth. Volume and mean depth were calculated from the measured dilution of a tritium addition. Enclosures were new except where noted and most received a ^{203}Hg addition.

Year	Designator		Treatment name	Bottom type	Volume (m^3)	Mean depth (m)
	Present	Previous				
1981	C81-1		Control	natural	234	2.98
	LM81-2		Low moss	natural	261	3.32
	PN81-3		Phosphorus & nitrogen	plastic	240	3.05
	LS81-4		Low sediment	plastic	305	3.89
	C81-5		Control	plastic	277	3.53
	HS81-6		High sediment	plastic	297	3.78
	C81-7		Control	plastic	276	3.52
	L81-8		Lake			
1982	FM82-1		Flooded moss-sedge-grass	natural	162	2.05
	LM82-3		Low moss	natural	267	3.40
	HM82-4		High moss	natural	200	2.55
	OC82-5	C81-1	Old control	natural	258	3.29
	C82-6		Control	natural	222	2.82
	HS82-7		High sediment	natural	217	2.76
	OM82-8	LM81-2	Old moss	natural	256	3.26
	L82-9		Lake			
1983	DFM83-1	FM82-1	Disturbed flooded moss	natural	184	2.34
	SP83-2		Black spruce	natural	261	3.33
	LM83-3		Low moss	natural	263	3.35
	OC83-4	C82-6	Old control	natural	253	3.22
	OAS83-5	HS82-7	Old added sediment	natural	235	2.99
	OM83-6	OM82-8, LM81-2	Old moss	natural	238	3.03
	LM83-7		Low moss	natural	235	2.99
	LMS83-8		Low moss and sediment	natural	275	3.50
	L83-9		Lake			
1984	LMN84-1		Low moss, no ^{203}Hg	natural	252	3.21
	CN84-2		Control, no ^{203}Hg	natural	272	3.47
	SP84-3		Black spruce	natural	289	3.68
	SD84-4		Prairie sod	natural	303	3.86
	LM84-5		Low moss	natural	247	3.15
	CHg84-6		Control, added Hg	natural	279	3.55
	C84-7		Control	natural	272	3.47
	L84-8		Lake			

Table 2. Type, mass, nutrient composition, total mercury content, time of addition, and mass of total mercury in treatment materials added to the enclosures. na = no analysis.

Limnocorral	Treatment	Time of addition	Mass added (kg)		Nutrient & Hg content ($mg \cdot g^{-1}$, dry wt)					Mass Hg added (g)
			Wet	Dry	C	N	P	CO_3	Hg	
LMB1-2	*Sphagnum* moss-peat	430 kg on June 21, remainder on June 24	565	63	419	3.9	0.4	na	0.10	6.3
PNB1-3	Phosphorus as $NaH_2PO_4 \cdot 2H_2O$ Nitrogen as $NaNO_3$	P: four equal additions every second week N: 4.1 g on June 21, remainder on July 15	- -	- -	- -	- 7.5[c]	3.7[b] -	- -	- -	- -
LSB1-4	Bank clay	Weekly to maintain Secchi near 0.5 m	43	33	24	0.3	0.6	16	0.02	0.66
HSB1-6	Bank clay	Weekly to maintain Secchi near 0.3 m	71	52	24	0.3	0.6	16	0.02	1.04
FMB2-1	Enclosure located over a land surface flooded in 1976	June 27	-	-	-	-	-	-	-	-
LMB2-3	Moss-peat	June 30	468	50	351	4.5	0.4	na	0.14	7.0
HMB2-4	Moss-peat	June 30	936	100	351	4.5	0.4	na	0.14	14.0
HSB2-7	Bank clay	Twice weekly to maintain Secchi near 0.35 m	74	50	17	0.5	0.6	13	0.04	2.0
DFMB3-1	Surface of moss-peat bed disturbed to about 5 cm depth	June 29	-	-	-	-	-	-	-	-
SPB3-2	Black spruce (*Picea mariana*) boughs <1 cm diameter	June 30	219	157						
LMB3-3	Moss-peat	June 30	414	47	385[c]	4.2[c]	0.4[c]	na	0.09	4.23
LMB3-7	Moss-peat	June 30	414	47	385[c]	4.2[c]	0.4[c]	na	0.09	4.23
LMSB3-8	Moss-peat and lake bottom sediment	Moss-peat on June 30 and sediment added twice weekly to maintain Secchi near 0.4 m	394 157	44	385[c]	4.2[c]	0.4[c]	na	0.09	3.96
LMNB4-1	Moss-peat	June 26	449	53	496	7.2	0.7	na	0.04	2.12
SPB4-3	Black spruce boughs <1 cm diameter	June 26	58	31	529	5.0	0.7	na	0.03	0.93
SDB4-4	Prairie sod	June 26	940	610	135	13.0	1.8	na	0.06	36.60
LMB4-5	Moss-peat	June 26	449	53	486	7.2	0.7	na	0.04	2.12
ChgB4-6	Mercury as $HgCl_2$	June 27	-	-	-	-	-	-	-	2.20×10^{-3}

[a,b] Total mass (g) added.

[c] Means of analyses on material added in 1981 and 1982.

Table 3. Total number of yellow perch and emerald shiners of given age class that were recovered/stocked in each limnocorral. The number of extraneous fish (i.e. those trapped during limnocorral construction) recovered is also shown.

Year	Limnocorral Designator	Fish recovered/stocked				Extraneous fish recovered
		Yellow perch			Emerald shiner	
		(0+)	(1+)	(2+)	(2+)	
1981	C81-1		23/111		12/28	207
	LM81-2		46/111		11/29	29
	PN81-3		20/112		4/28	4
	LS81-4		34/108		8/27	23
	C81-5		34/110		8/29	0
	HS81-6		33/111		13/27	2
	C81-7		28/111		12/30	4
1982	FM82-1		64/80			36
	LM82-3		80/80			89
	HM82-4		62/80			27
	OC82-5		29/80			21
	C82-6		79/80			27
	HS82-7		71/80			74
	OM82-8		65/80			22
1983	DFM83-1	14/60		27/32		52
	SP83-2	0/0		8/32		5
	LM83-3	26/60		27/32		1
	OC83-4	31/60		20/32		9
	OAS83-5	39/60		32/32		63
	OM83-6	0/60		4/32		7
	LM83-7	58/60		31/32		35
	LMS83-8	60/60		32/32		73
1984	LMN84-1		100/100			62
	CN84-2		85/100			22
	SP84-3		87/100			29
	SD84-4		84/100			57
	LM84-5		68/100			26
	CHg84-6		77/100			2
	C84-7		86/100			1

Table 4. Growth of yellow perch and emerald shiners in limnocorrals, 1981-84. Data for the Aug 29, 1984 sample has been included because the 1981-83 experiments all ended at approximately this time. Age of fish used each year is shown in parentheses. The 0+ perch in 1983 were added Aug 5 (24 days in limnocorral). (n/s = no sample).

Year	Limnocorral designator	Initial round wt - final round wt (g)				Gain in round wt (g)				Duration of experiment	Average wt gain (g/day)			
		Yellow perch			Emerald shiner	Yellow perch			Emerald shiner		Yellow perch			Emerald shiner
		(0+)	(1+)	(2+)	(2+)	(0+)	(1+)	(2+)	(2+)		(0+)	(1+)	(2+)	(2+)
1981	C81-1		1.03-6.37		2.60-4.02		5.34		1.42	July 4-Aug 25 (52 days)		0.10		0.03
	LM81-2		1.03-5.56		2.60-4.28		4.53		1.68			0.09		0.03
	PN81-3		1.03-n/s		2.60-4.00		n/s		1.40			n/s		0.03
	LS81-4		1.03-5.13		2.60-4.00		4.10		1.40			0.08		0.03
	C81-5		1.03-5.77		2.60-3.89		4.74		1.29			0.09		0.02
	HS81-6		1.03-6.30		2.60-3.85		5.27		1.25			0.10		0.02
	C81-7		1.03-n/s		2.60-3.94		n/s		1.34			n/s		0.03
1982	FM82-1		1.43-3.44				2.01			July 4-Aug 30 (57 days)		0.04		
	LM82-3		1.43-3.45				2.02					0.04		
	HM82-4		1.43-4.81				3.38					0.06		
	OC82-5		1.43-3.97				2.54					0.04		
	C82-6		1.43-3.52				2.09					0.04		
	HS82-7		1.43-2.96				1.53					0.03		
	OM82-8		1.43-4.17				2.74					0.05		
1983	DFM83-1	0.38-1.60		3.60-7.79		1.22		4.19		July 7-Aug 29 (53 days)	0.05		0.08	
	SP83-2	0.38-n/s		3.60-n/s		n/s		n/s			n/s		n/s	
	LM83-3	0.38-1.17		3.60-7.52		0.79		3.92			0.03		0.07	
	OC83-4	0.38-1.33		3.60-8.45		0.95		4.85			0.04		0.09	
	OAS83-5	0.38-1.57		3.60-7.75		1.19		4.15			0.05		0.08	
	OM83-6	0.38-n/s		3.60-n/s		n/s		n/s			n/s		n/s	
	LM83-7	0.38-1.23		3.60-6.17		0.85		2.57			0.04		0.05	
	LMS83-8	0.38-1.07		3.60-7.05		0.69		3.45			0.03		0.06	
1984	LM84-1		1.72-3.06				1.34			June 27-Aug 29 (63 days)		0.05		
	CN84-2		1.72-3.77				2.05					0.06		
	SP84-3		1.72-3.63				1.91					0.06		
	SD84-4		1.72-7.60				5.88					0.12		
	LM84-5		1.72-3.88				2.16					0.06		
	CHg84-6		1.72-2.49				0.77					0.04		
	C84-7		1.72-n/s				n/s					n/s		
	LMN84-1		1.72-3.48				1.76			June 27-Sept 30 (95 days)		0.02		
	CN84-2		1.72-2.85				1.13					0.01		
	SP84-3		1.72-4.79				3.07					0.03		
	SD84-4		1.72-7.11				5.39					0.06		
	LM84-5		1.72-4.19				2.47					0.03		
	CHg84-6		1.72-4.86				3.14					0.03		
	C84-7		1.72-4.41				2.69					0.03		

Table 5. Activity of ^{3}H and ^{203}Hg spikes added to the limnocorrals, concentration in water following the spike, measured concentration of total mercury in the water after the spike, total mercury added in the spikes, concentration based on specific activity, and estimated maximum contribution of spike to water mercury concentration based on specific activity, 1981-1984.

Year	Limnocorral	Activity added (Bq)		Concentration in water ($Bq \cdot L^{-1}$)		Specific activity of ^{203}Hg spike	Mass Hg Added (mg)	Total Hg in water ($ng \cdot L^{-1}$)	
		^{3}H ($x10^{9}$)	^{203}Hg ($x10^{8}$)	^{3}H ($x10^{4}$)	^{203}Hg ($x10^{2}$)[a]	($x10^{8}$ Bq (mg Hg)$^{-1}$)		Measured	Due to spike[b]
1981	C81-1	4.63	1.55	1.98	6.63	0.85	1.85	10	8
	LM81-2	4.63	1.55	1.77	5.94		1.82	<5	7
	PN81-3	4.63	1.32	1.93	5.51		1.55	<5	7
	LS81-4	4.63	1.55	1.52	5.08		1.82	<5	6
	C81-5	4.63	1.55	1.67	5.60		1.82	7	7
	HS81-6	4.63	1.55	1.56	5.23		1.82	<5	3
	C81-7	4.63	1.55	1.68	5.61		1.82	<5	7
	L81-8							<5	
1982	FM82-1	3.70	1.39	2.28	8.60	1.22	1.14	20	7
	LM82-3	3.70	1.39	1.39	5.21		1.14	<6	4
	HM82-4	3.70	1.39	1.85	6.94		1.14	16	6
	OC82-5	3.70	1.39	1.43	5.38		1.14	11	4
	C82-6	3.70	1.39	1.67	6.26		1.14	11	5
	HS82-7	3.70	1.39	1.70	6.40		1.14	9	5
	OM82-8	3.70	1.39	1.45	5.43		1.14	8	4
	L82-9							<2	
1983	DFM83-1	3.70	1.86	2.02	10.13	1.35	1.38	na	8
	SP83-2	3.70	1.86	1.42	7.12		1.38		5
	LM83-3	3.70	1.86	1.41	7.07		1.38		5
	OC83-4	3.70	1.86	1.47	7.36		1.38		5
	OAS83-5	3.70	1.86	1.58	7.92		1.38		6
	OM83-6	3.70	1.86	1.56	7.82		1.38		6
	LM83-7	3.70	1.86	1.58	7.92		1.38		6
	LMS83-8	3.70	1.86	1.35	6.77		1.38		5
1984	SP84-3	0.24	1.39	0.08	4.81	2.11	0.66	na	2
	SD84-4	0.24	1.39	0.08	4.58		0.66		2
	LM84-5	0.24	1.39	0.10	5.63		0.66		3
	CHg84-6	0.24	1.39	0.09	5.10		0.66		2
	C84-7	0.24	1.39	0.09	5.10		0.66		2

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[a] Nominal concentration estimated from activity added and volume of limnocorral.

[b] Nominal concentration estimated from specific activity of spike and volume of limnocorral.

na No analysis.

Table 6. Activity of other isotopes added to the limnocorrals in 1981 and nominal concentration in the water immediately following the spike. ^{22}Na was added only to LS81-4.

Parameter	Limnocorral	Isotope ^{75}Se	 ^{134}Cs	 ^{65}Zn	 ^{60}Co	 ^{22}Na
Activity added ($\times 10^7$ Bq)	all	3.29	8.33	39.0	22.0	3.70
Concentration in water ($\times 10^2$ Bq$\cdot$L^{-1})	C81-1	1.41	3.56	16.7	9.40	-
	LM81-2	1.26	3.19	14.9	8.43	-
	PN81-3	1.37	3.47	16.3	9.17	-
	LS81-4	1.08	2.73	12.8	7.21	1.21
	C81-5	1.19	3.01	14.1	7.94	-
	HS81-6	1.11	2.80	13.1	7.41	-
	C81-7	1.19	3.02	14.1	7.97	-

Table 7. Comparison of mean (± SD) ^{203}Hg activity (cpm·g^{-1}, wet wt) in fish at the end of the 1981 experiment. (Standard deviation not included when n=1). Values presented for the stocked perch and shiners are for mathematically combined muscle and carcass counts. Values for the extraneous fish represent counts of whole fish.

Species	^{203}Hg activity						
	C81-1	LM81-2	PN81-3	LS81-4	C81-5	HS81-6	C81-7
Stocked fish:							
Yellow perch	456 (18)	20,657 (3693)		2,015 (646)	1,220 (235)	1,169 (67)	
Emerald shiner shiner	602 (90)	20,234 (3145)	1,450 (181)	2,715 (422)	1,561 (156)	1,519 (225)	1,010 (251)
Extraneous fish:							
Stickleback	578 (226)						
Trout-perch	936 (398)	13,178					
Cisco	2,073						
Burbot		8,390					
Spottail shiner		25,842 (6529)	2,607				
Johnny darter		31,351 (10,525)					
Sculpin				2,428		2,990	
Emerald shiner				1,040			

Table 8. Mean percentage (± SD) of ^{203}Hg in fish muscle present as methyl mercury in the 1981 limnocorral experiment. (n/s = no sample).

Limnocorral	Yellow perch		Emerald shiners
	August 2-3	August 24-25	August 24-25
C81-1	91 (8)	92 (17)	118 (8)
LM81-2	99 (3)	97 (14)	102 (5)
PN81-3	79 (13)	n/s	92 (3)
LS81-4	87 (8)	100 (9)	103 (5)
C81-5	61 (9)	79 (7)	79 (2)
HS81-6	95 (22)	100 (1)	94 (2)
C81-7	44 (15)	n/s	50 (12)

Table 9. Mean concentration (± SD) of total mercury ($\mu g \cdot g^{-1}$, wet wt) in yellow perch muscle at the start, middle, and end of the 1981 experiment and specific activity ($x10^8$ $Bq \cdot (mg \cdot Hg)^{-1}$) of ^{203}Hg in perch muscle at the end of the experiment. (n/s = no sample).

Limnocorral	Total Hg concentration			Specific activity
	July 1	August 2-3	August 24-25	August 24-25
C81-1	0.090 (0.020)	0.034 (0.023)	n/s	n/s
LM81-2	0.090 (0.020)	0.200 (0.030)	0.470 (0.080)	0.210 (0.023)
PN81-3	0.090 (0.020)	0.041 (0.005)	n/s	n/s
LS81-4	0.090 (0.020)	0.044 (0.007)	0.039 (0.008)	0.230 (0.043)
C81-5	0.090 (0.020)	0.039 (0.012)	0.065 (0.027)	0.070 (0.024)
HS81-6	0.090 (0.020)	0.040 (0.005)	n/s	n/s
C81-7	0.090 (0.020)	0.053 (0.018)	n/s	n/s

Table 10. Comparison of the specific activity of the ^{203}Hg spike (corrected for the mass of Hg added in the treatment materials) to the specific activity of ^{203}Hg in fish muscle at the end of the 1984 experiment. The corrected specific activity of the spike assumes that all of the Hg added in the treatments was available.

Limnocorral	Mass Hg added (mg)		Corrected specific activity of spike (Bq·(mg Hg)$^{-1}$)	Specific activity in fish muscle (Bq·(mg Hg)$^{-1}$)
	Spike	Treatments		
SP84-3	0.66	9.3×10^2	1.5×10^5	3.6×10^7
SD84-4	0.66	3.7×10^4	3.7×10^3	2.9×10^7
LM84-5	0.66	2.1×10^3	6.6×10^4	1.6×10^7
CHg84-6	0.66	2.2	4.9×10^7	5.6×10^6
C84-7	0.66	-	2.1×10^8	8.4×10^6

Table 11. Comparison of seasonal mean Secchi disk visibility, chlorophyll a concentration, and water column temperature in Wupaw Bay with growth of yellow perch in the control limnocorrals, 1981-1984. Fish growth represents the increase in mean round weight of added yellow perch over the course of the experiment.

	1981	1982	1983	1984
Temperature (°C)	18.59	16.98	19.05	19.48
Secchi (m)	1.46	1.28	1.56	1.40
Chlorophyll a ($\mu g \cdot L^{-1}$)	2.78	2.00	2.12	NA
Perch growth (g)	5.34	2.09	4.85*	2.05**

* This is the weight gain of 2+ yellow perch; values for other years are for 1+ perch. The weight gain of 1+ perch in 1983 may have been significantly less than 4.85 g

** This is the weight gain from CN84-2 from the beginning of the experiment to Aug 29, which is approximately the same period as in 1981-83.

Appendix 1

Summary of temperature, conductivity, and Secchi disc visibility measurements in the limnocorrals and surrounding bay, 1981-1984

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Table A.1.1. Summary of physical conditions in the limnocorrals, 1981.

Limnocorral	Date	Day of year	Secchi (m)	Mean water column	
				Temp. (C)	Cond. (µS/cm)
C81-1	20-Jun	171	1.75	-	-
	23-Jun	174	1.70	-	-
	26-Jun	177	1.60	-	-
	01-Jul	182	2.30	18.66	67
	02-Jul	183	1.60	-	-
	03-Jul	184	2.10	-	-
	07-Jul	188	1.90	17.86	66
	10-Jul	191	1.50	17.22	-
	13-Jul	194	1.90	19.14	70
	19-Jul	200	1.90	18.95	68
	27-Jul	208	1.35	18.48	71
	02-Aug	214	1.90	17.99	73
	03-Aug	215	1.50	-	-
	13-Aug	225	1.79	18.64	74
	14-Aug	226	1.90	-	-
	16-Aug	228	1.70	-	-
	19-Aug	231	-	19.60	-
	20-Aug	232	-	18.82	-
	21-Aug	233	2.10	-	-
	23-Aug	235	2.60	21.15	-
LM81-2	20-Jun	171	1.90	-	-
	24-Jun	175	1.85	-	-
	26-Jun	177	1.50	-	-
	01-Jul	182	2.25	18.69	56
	02-Jul	183	1.70	-	-
	03-Jul	184	2.10	-	-
	07-Jul	188	2.10	17.97	54
	10-Jul	191	1.40	16.98	-
	13-Jul	194	1.80	19.38	57
	19-Jul	200	2.50	19.05	57
	27-Jul	208	2.23	18.43	59
	02-Aug	214	3.00	18.21	61
	03-Aug	215	3.00	-	-
	13-Aug	225	2.95	19.02	60
	14-Aug	226	2.90	-	-
	16-Aug	228	2.80	-	-
	19-Aug	231	-	19.47	-
	20-Aug	232	-	18.93	-
	21-Aug	233	2.50	-	-
	23-Aug	235	2.80	21.37	-
PN81-3	20-Jun	171	1.90	-	-
	23-Jun	174	2.15	-	-
	26-Jun	177	2.10	-	-
	01-Jul	182	2.30	18.91	68
	02-Jul	183	1.60	-	-
	03-Jul	184	1.80	-	-
	07-Jul	188	1.60	18.01	65
	10-Jul	191	1.20	17.07	-
	13-Jul	194	1.50	19.23	69
	19-Jul	200	1.20	19.01	68
	27-Jul	208	1.03	18.64	69
	02-Aug	214	1.10	18.13	70
	03-Aug	215	0.90	-	-
	13-Aug	225	1.23	18.60	70
	14-Aug	226	1.40	-	-
	16-Aug	228	1.30	-	-
	19-Aug	231	-	19.54	-

Table A.1.1. Cont'd.

Limnocorral	Date	Day of year	Secchi (m)	Mean water column Temp. (C)	Mean water column Cond. (µS/cm)
	20-Aug	232	-	19.01	-
	21-Aug	233	1.30	-	-
	23-Aug	235	1.20	21.29	-
LS81-4	20-Jun	171	1.90	-	-
	23-Jun	174	0.62	-	-
	24-Jun	175	0.67	-	-
	26-Jun	177	1.10	-	-
	01-Jul	182	1.80	18.89	73
	02-Jul	183	0.80	-	-
	03-Jul	184	1.00	-	-
	07-Jul	188	1.40	18.10	71
	08-Jul	189	0.50	-	-
	10-Jul	191	0.55	16.94	-
	13-Jul	194	1.30	18.68	77
	15-Jul	196	1.50	-	-
	15-Jul	196	0.50	-	-
	19-Jul	200	0.90	19.16	76
	22-Jul	203	1.10	-	-
	22-Jul	203	0.40	-	-
	27-Jul	208	0.65	18.66	79
	30-Jul	211	0.45	-	-
	30-Jul	211	0.80	-	-
	02-Aug	214	0.55	18.03	81
	03-Aug	215	0.50	-	-
	04-Aug	216	0.70	-	-
	06-Aug	218	0.50	-	-
	06-Aug	218	0.80	-	-
	13-Aug	225	0.50	18.76	85
	13-Aug	225	1.00	-	-
	14-Aug	226	0.60	-	-
	16-Aug	228	0.65	-	-
	19-Aug	231	-	19.39	-
	20-Aug	232	0.80	18.76	-
	20-Aug	232	0.50	-	-
	21-Aug	233	0.60	-	-
	23-Aug	235	0.60	21.31	-
C81-5	20-Jun	171	2.00	-	-
	23-Jun	174	2.25	-	-
	26-Jun	177	1.90	-	-
	01-Jul	182	2.50	19.37	69
	02-Jul	183	2.10	-	-
	03-Jul	184	2.50	-	-
	07-Jul	188	2.10	18.15	65
	10-Jul	191	1.95	16.79	-
	13-Jul	194	2.10	19.12	68
	19-Jul	200	2.20	19.21	67
	27-Jul	208	-	18.79	68
	02-Aug	214	2.05	18.01	67
	03-Aug	215	1.90	-	-
	13-Aug	225	3.05	18.81	68
	14-Aug	226	3.20	-	-
	16-Aug	228	3.30	-	-
	19-Aug	231	-	19.44	-
	20-Aug	232	-	18.89	-
	21-Aug	233	3.20	-	-
	23-Aug	235	3.40	21.00	-

Table A.1.1. Cont'd.

				Mean water column	
Limnocorral	Date	Day of year	Secchi (m)	Temp. (C)	Cond. (µS/cm)
HS81-6	20-Jun	171	2.30	-	-
	23-Jun	174	0.30	-	-
	24-Jun	175	0.35	-	-
	26-Jun	177	0.40	-	-
	01-Jul	182	0.80	18.74	75
	02-Jul	183	0.30	-	-
	03-Jul	184	0.35	-	-
	07-Jul	188	0.60	18.10	74
	08-Jul	189	0.30	-	-
	10-Jul	191	0.39	16.75	-
	13-Jul	194	0.70	19.16	79
	15-Jul	196	1.00	-	-
	15-Jul	196	0.30	-	-
	19-Jul	200	0.60	19.29	82
	22-Jul	203	0.80	-	-
	22-Jul	203	0.30	-	-
	27-Jul	208	0.60	18.95	87
	30-Jul	211	0.60	-	-
	30-Jul	211	0.30	-	-
	02-Aug	214	0.50	18.13	92
	03-Aug	215	0.50	-	-
	04-Aug	216	0.55	-	-
	06-Aug	218	0.35	-	-
	06-Aug	218	0.60	-	-
	13-Aug	225	0.35	18.42	93
	13-Aug	225	0.60	-	-
	14-Aug	226	0.40	-	-
	16-Aug	228	0.50	-	-
	19-Aug	231	-	19.37	-
	20-Aug	232	0.65	18.70	-
	20-Aug	232	0.35	-	-
	21-Aug	233	0.40	-	-
	23-Aug	235	0.35	20.46	-
	23-Aug	235	0.45	-	-
C81-7	20-Jun	171	2.10	-	-
	23-Jun	174	2.60	16.25	-
	26-Jun	177	2.00	-	-
	01-Jul	182	2.80	19.67	68
	02-Jul	183	2.75	-	-
	07-Jul	188	2.00	18.14	65
	10-Jul	191	1.90	16.80	68
	13-Jul	194	2.00	18.65	66
	19-Jul	200	2.00	19.73	68
	27-Jul	208	2.05	18.74	67
	02-Aug	214	2.35	18.01	67
	03-Aug	215	2.50	-	-
	13-Aug	225	2.80	-	66
	14-Aug	226	3.00	-	-
	16-Aug	228	3.00	-	-
	19-Aug	231	-	19.36	-
	20-Aug	232	-	18.64	-
	21-Aug	233	3.00	-	-
	23-Aug	235	3.00	20.01	-
L81-8	20-Jun	171	1.30	-	-
	24-Jun	175	1.50	-	-
	26-Jun	177	1.45	-	-
	01-Jul	182	1.80	19.09	72
	02-Jul	183	1.30	-	-

Table A.1.1. Cont'd.

Limnocorral	Date	Day of year	Secchi (m)	Mean water column	
				Temp. (C)	Cond. (µS/cm)
	03-Jul	184	1.10	-	-
	07-Jul	188	1.30	17.98	67
	10-Jul	191	0.88	16.74	-
	13-Jul	194	1.40	19.40	-
	19-Jul	200	1.70	18.97	72
	27-Jul	208	1.72	18.41	76
	02-Aug	214	1.70	18.25	75
	03-Aug	215	1.70	-	-
	13-Aug	225	1.10	18.66	77
	16-Aug	228	1.70	-	-
	19-Aug	231	-	19.90	-
	20-Aug	232	-	19.41	-
	21-Aug	233	1.80	-	-
	23-Aug	235	1.55	20.53	-

Table A.1.2. Summary of physical conditions in the limnocorrals, 1982.

Limnocorral	Date	Day of year	Secchi (m)	Mean water column	
				Temp. (C)	Cond. (µS/cm)
FM82-1	29-Jun	180	1.00	14.27	84
	04-Jul	185	1.60	-	-
	07-Jul	188	1.40	16.61	86
	08-Jul	189	1.50	-	-
	10-Jul	191	1.50	-	-
	12-Jul	193	1.80	-	-
	14-Jul	195	1.65	18.11	86
	15-Jul	196	2.20	-	-
	21-Jul	202	1.75	17.04	87
	26-Jul	207	2.45	-	-
	28-Jul	209	2.35	18.86	87
	02-Aug	214	2.40	-	-
	04-Aug	216	1.90	13.92	84
	06-Aug	218	1.80	-	-
	09-Aug	221	1.80	15.99	-
	13-Aug	225	1.70	-	-
	16-Aug	228	1.80	-	-
	20-Aug	232	1.80	17.70	86
	23-Aug	235	1.60	15.80	68
	26-Aug	238	1.65	12.42	81
	30-Aug	242	1.75	-	-
SC82-2	29-Jun	180	1.20	14.35	84
	04-Jul	185	-	18.01	-
	07-Jul	188	1.55	16.30	85
	08-Jul	189	1.70	-	-
	12-Jul	193	2.10	-	-
	14-Jul	195	1.90	16.92	86
	15-Jul	196	2.20	-	-
	19-Jul	200	2.20	-	-
	21-Jul	202	2.20	16.94	86
	26-Jul	207	2.50	-	-
	28-Jul	209	2.70	18.56	86
	29-Jul	210	2.70	-	-
	31-Jul	212	2.70	-	-
	02-Aug	214	2.40	-	-
	02-Aug	214	2.40	-	-
	03-Aug	215	2.60	15.88	82
	06-Aug	218	-	16.58	-
	09-Aug	221	2.90	15.59	-
	13-Aug	225	2.90	-	-
	16-Aug	228	1.90	-	-
	20-Aug	232	3.00	17.31	85
	23-Aug	235	2.95	16.13	84
	26-Aug	238	2.40	13.13	79
	30-Aug	242	2.80	-	-
LM82-3	29-Jun	180	1.00	13.85	84
	04-Jul	185	1.60	17.81	-
	07-Jul	188	1.90	16.00	67
	08-Jul	189	2.00	-	-
	10-Jul	191	2.00	-	-
	12-Jul	193	2.85	-	-
	14-Jul	195	2.30	17.49	70
	15-Jul	196	2.30	-	-
	19-Jul	200	2.40	-	-
	21-Jul	202	2.70	16.77	77
	26-Jul	207	3.50	-	-
	28-Jul	209	2.70	19.03	71
	31-Jul	212	3.00	18.77	71
	01-Aug	213	2.95	-	69
	02-Aug	214	2.80	-	-
	03-Aug	215	3.00	15.53	66
	04-Aug	216	2.60	14.50	67

Table A.1.2. Cont'd.

Limnocorral	Date	Day of year	Secchi (m)	Mean water column	
				Temp. (C)	Cond. (µS/cm)
	06-Aug	218	2.60	-	-
	09-Aug	221	2.80	16.24	-
	09-Aug	221	-	17.03	-
	13-Aug	225	2.70	-	-
	16-Aug	228	2.10	-	-
	20-Aug	232	1.80	17.60	73
	23-Aug	235	1.70	16.07	70
	26-Aug	238	2.00	12.80	67
	30-Aug	242	2.00	-	-
HM82-4	29-Jun	180	1.05	15.16	85
	04-Jul	185	-	17.27	-
	07-Jul	188	1.70	16.19	53
	08-Jul	189	1.90	-	-
	10-Jul	191	2.10	-	-
	12-Jul	193	2.50	-	-
	14-Jul	195	1.70	17.86	52
	15-Jul	196	1.90	-	-
	19-Jul	200	2.30	-	-
	21-Jul	202	2.10	16.83	51
	26-Jul	207	2.05	-	-
	28-Jul	209	2.80	18.65	55
	02-Aug	214	1.65	-	-
	04-Aug	216	1.80	14.20	47
	06-Aug	218	1.70	-	-
	09-Aug	221	1.65	16.32	-
	13-Aug	225	1.65	-	-
	16-Aug	228	1.70	-	-
	20-Aug	232	1.45	17.65	54
	23-Aug	235	1.30	15.89	52
	26-Aug	238	1.90	12.58	49
	30-Aug	242	1.85	-	-
OC82-5	29-Jun	180	1.15	13.71	84
	04-Jul	185	1.60	17.48	-
	07-Jul	188	1.70	16.34	86
	08-Jul	189	1.70	-	-
	10-Jul	191	2.00	-	-
	12-Jul	193	2.20	-	-
	14-Jul	195	2.10	18.13	87
	15-Jul	196	2.20	-	-
	19-Jul	200	2.20	-	-
	21-Jul	202	2.40	16.83	87
	26-Jul	207	2.90	-	-
	28-Jul	209	2.70	18.95	88
	02-Aug	214	2.50	-	-
	03-Aug	215	2.60	15.34	82
	06-Aug	218	2.30	-	-
	09-Aug	221	2.75	16.40	-
	13-Aug	225	2.70	-	-
	16-Aug	228	2.50	-	-
	20-Aug	232	2.65	17.66	87
	23-Aug	235	2.50	15.98	86
	26-Aug	238	2.10	12.72	81
	30-Aug	242	2.15	-	-
C82-6	29-Jun	180	1.20	15.03	85
	04-Jul	185	-	17.57	-
	07-Jul	188	1.60	16.36	85
	08-Jul	189	1.80	-	-
	10-Jul	191	2.00	-	-
	12-Jul	193	2.10	-	-
	14-Jul	195	2.20	18.28	88
	15-Jul	196	2.20	-	-
	19-Jul	200	2.30	-	-

Table A.1.2. Cont'd.

Limnocorral	Date	Day of year	Secchi (m)	Mean water column	
				Temp. (C)	Cond. (µS/cm)
	21-Jul	202	2.15	17.13	88
	26-Jul	207	2.90	-	-
	28-Jul	209	2.50	18.85	88
	02-Aug	214	2.00	-	-
	06-Aug	218	1.90	-	-
	09-Aug	221	2.90	16.27	-
	13-Aug	225	2.10	-	-
	16-Aug	228	2.20	-	-
	20-Aug	232	2.05	17.57	88
	23-Aug	235	1.90	15.99	85
	26-Aug	238	1.80	12.76	80
	30-Aug	242	2.05	-	-
HS82-7	29-Jun	180	1.15	14.30	84
	02-Jul	183	1.10	-	-
	02-Jul	183	0.35	-	-
	04-Jul	185	0.45	17.50	-
	05-Jul	186	0.33	-	-
	05-Jul	186	0.55	-	-
	07-Jul	188	0.40	16.20	97
	08-Jul	189	0.30	-	-
	08-Jul	189	0.35	-	-
	08-Jul	189	0.50	-	-
	10-Jul	191	0.40	-	-
	12-Jul	193	0.37	-	-
	12-Jul	193	0.40	-	-
	12-Jul	193	0.33	-	-
	14-Jul	195	0.55	18.23	105
	15-Jul	196	0.60	-	-
	15-Jul	196	0.45	-	-
	15-Jul	196	0.45	-	-
	19-Jul	200	1.00	-	-
	19-Jul	200	1.00	-	-
	19-Jul	200	0.35	-	-
	21-Jul	202	0.40	16.72	110
	22-Jul	203	0.40	-	-
	22-Jul	203	0.47	-	-
	26-Jul	207	0.40	-	-
	28-Jul	209	0.55	18.56	113
	29-Jul	210	0.55	-	-
	29-Jul	210	0.39	-	-
	02-Aug	214	0.35	-	-
	02-Aug	214	0.60	-	-
	03-Aug	215	0.40	15.26	109
	06-Aug	218	0.40	-	-
	06-Aug	218	0.55	-	-
	09-Aug	221	0.60	-	-
	09-Aug	221	0.35	15.94	-
	13-Aug	225	0.65	-	-
	13-Aug	225	0.38	-	-
	16-Aug	228	0.80	-	-
	16-Aug	228	0.30	-	-
	20-Aug	232	0.65	17.51	124
	20-Aug	232	0.39	-	124
	23-Aug	235	0.50	15.92	123
	25-Aug	237	0.60	-	-
	25-Aug	237	0.30	-	-
	26-Aug	238	0.35	12.66	114
	30-Aug	242	0.45	-	-
OM82-8	29-Jun	180	1.30	14.30	84
	04-Jul	185	1.80	17.55	-
	07-Jul	188	1.65	16.44	85
	08-Jul	189	2.10	-	-
	10-Jul	191	2.20	-	-

Table A.1.2. Cont'd.

Limnocorral	Date	Day of year	Secchi (m)	Mean water column Temp. (C)	Mean water column Cond. (µS/cm)
	12-Jul	193	3.20		-
	14-Jul	195	2.20	18.05	86
	15-Jul	196	3.00	-	-
	19-Jul	200	2.90	-	-
	21-Jul	202	2.30	17.17	85
	26-Jul	207	3.50	-	-
	28-Jul	209	3.50	18.56	86
	02-Aug	214	2.55	-	-
	03-Aug	215	3.25	15.43	78
	06-Aug	218	2.60	-	-
	09-Aug	221	2.70	16.24	-
	13-Aug	225	2.35	-	-
	16-Aug	228	2.00	-	-
	20-Aug	232	2.10	17.81	83
	23-Aug	235	2.00	16.10	81
	26-Aug	238	1.95	13.03	74
	30-Aug	242	2.30	-	-
L82-9	29-Jun	180	1.00	17.05	86
	04-Jul	185	1.20	-	-
	07-Jul	188	-	15.81	85
	08-Jul	189	0.95	-	-
	10-Jul	191	1.25	-	-
	14-Jul	195	1.30	18.21	88
	15-Jul	196	1.50	-	-
	19-Jul	200	1.25	-	-
	21-Jul	202	0.90	16.53	88
	26-Jul	207	1.30	-	-
	28-Jul	209	1.30	19.14	90
	02-Aug	214	1.60	-	-
	03-Aug	215	1.20	15.89	86
	06-Aug	218	1.80	-	-
	09-Aug	221	1.40	15.87	-
	13-Aug	225	1.40	-	-
	20-Aug	232	1.40	17.88	94
	23-Aug	235	1.40	15.93	91
	26-Aug	238	1.20	12.85	83
	30-Aug	242	1.30	-	-

Table A.1.3. Summary of physical conditions in the limnocorrals, 1983.

				Mean water column		
Limnocorral	Date	Day of year	Secchi (m)	Temp. (C)	Cond. (µS/cm)	Dissolved oxygen (mg/L)
DFM83-1	23-Jun	175	1.00	16.53	83	8.81
	30-Jun	182	0.50	16.85	83	7.04
	05-Jul	187	1.40	16.02	85	7.77
	07-Jul	189	1.60	-	-	-
	09-Jul	191	1.60	-	-	-
	10-Jul	192	1.60	-	-	-
	12-Jul	194	1.50	19.76	87	7.97
	14-Jul	196	1.25	-	-	-
	18-Jul	200	1.80	-	-	-
	20-Jul	202	1.90	19.72	88	7.08
	24-Jul	206	2.00	-	-	-
	26-Jul	208	1.75	-	-	-
	28-Jul	210	1.70	15.22	88	6.57
	31-Jul	213	1.80	-	-	-
	03-Aug	216	1.50	-	-	-
	04-Aug	217	1.20	21.21	91	6.39
	10-Aug	223	1.20	-	-	-
	11-Aug	224	1.30	21.13	92	5.91
	14-Aug	227	1.40	-	-	-
	19-Aug	232	1.00	-	-	-
	21-Aug	234	1.00	16.62	88	7.80
	22-Aug	235	-	17.45	88	8.86
	23-Aug	236	-	16.50	88	7.99
	25-Aug	238	0.80	18.08	89	8.31
SP83-2	23-Jun	175	1.20	16.55	84	8.67
	30-Jun	182	2.00	17.10	83	7.67
	05-Jul	187	1.10	16.06	86	5.36
	07-Jul	189	1.00	-	-	-
	09-Jul	191	1.05	-	-	-
	10-Jul	192	0.95	-	-	-
	12-Jul	194	0.75	19.61	91	1.37
	14-Jul	196	0.70	-	-	-
	18-Jul	200	0.80	-	-	-
	20-Jul	202	0.60	20.02	93	0.22
	24-Jul	206	0.50	-	-	-
	26-Jul	208	0.40	-	-	-
	28-Jul	210	0.60	20.50	96	0.28
	31-Jul	213	0.60	-	-	-
	03-Aug	216	0.40	-	-	-
	04-Aug	217	0.20	21.08	101	0.51
	10-Aug	223	0.40	-	-	-
	11-Aug	224	0.35	21.00	102	0.69
	14-Aug	227	0.45	-	-	-
	19-Aug	232	0.30	-	-	-
	21-Aug	234	-	16.72	99	0.62
	22-Aug	235	-	17.47	99	1.99
	23-Aug	236	-	16.40	98	0.48
	25-Aug	238	0.30	17.75	102	1.13
LM83-3	23-Jun	175	1.00	16.44	85	8.65
	30-Jun	182	1.60	16.89	73	7.90
	05-Jul	187	1.90	16.09	68	7.70
	07-Jul	189	2.20	-	-	-
	09-Jul	191	2.40	-	-	-
	10-Jul	192	2.10	-	-	-
	12-Jul	194	2.00	19.70	70	7.70
	14-Jul	196	1.90	-	-	-

Table A.1.3. Cont'd.

				Mean water column		
Limnocorral	Date	Day of year	Secchi (m)	Temp. (C)	Cond. (µS/cm)	Dissolved oxygen (mg/L)
	17-Jul	199	2.30	-	-	-
	20-Jul	202	1.90	19.64	72	6.80
	24-Jul	206	2.10	-	-	-
	26-Jul	208	1.80	-	-	-
	28-Jul	210	2.10	19.03	74	6.10
	31-Jul	213	2.20	-	-	-
	03-Aug	216	2.40	-	-	-
	04-Aug	217	2.40	19.35	77	6.57
	10-Aug	223	1.80	-	-	-
	11-Aug	224	2.30	21.09	79	6.92
	14-Aug	227	2.00	-	-	-
	19-Aug	232	2.00	-	-	-
	21-Aug	234	2.50	16.66	78	7.61
	22-Aug	235	-	17.36	78	8.14
	23-Aug	236	-	16.68	78	7.47
	25-Aug	238	2.30	18.07	80	8.31
OC83-4	23-Jun	175	1.00	16.66	84	8.43
	30-Jun	182	1.80	17.20	85	8.54
	05-Jul	187	1.80	15.97	86	8.50
	07-Jul	189	2.00	-	-	-
	09-Jul	191	2.30	-	-	-
	10-Jul	192	2.00	-	-	-
	12-Jul	194	2.10	19.78	88	8.65
	14-Jul	196	2.20	-	-	-
	17-Jul	199	1.90	-	-	-
	20-Jul	202	2.00	19.61	90	7.72
	24-Jul	206	2.00	-	-	-
	26-Jul	208	2.05	-	-	-
	28-Jul	210	2.20	19.09	89	7.44
	31-Jul	213	2.20	-	-	-
	03-Aug	216	2.90	-	-	-
	04-Aug	217	2.40	21.28	92	7.45
	10-Aug	223	2.20	-	-	-
	11-Aug	224	2.20	20.86	92	7.75
	13-Aug	226	2.40	-	-	-
	14-Aug	227	2.20	-	-	-
	19-Aug	232	2.00	-	-	-
	21-Aug	234	2.25	16.62	92	7.70
	22-Aug	235	-	17.36	91	7.85
	23-Aug	236	-	16.60	91	7.22
	25-Aug	238	2.00	18.03	93	8.81
SA83-5	23-Jun	175	1.20	16.41	84	8.90
	30-Jun	182	1.80	17.33	85	8.61
	05-Jul	187	2.00	15.94	87	8.92
	07-Jul	189	2.10	-	-	-
	09-Jul	191	2.40	-	-	-
	10-Jul	192	2.40	-	-	-
	12-Jul	194	2.60	19.62	91	8.49
	14-Jul	196	2.70	-	-	-
	18-Jul	200	2.60	-	-	-
	20-Jul	202	2.95	19.63	92	8.35
	24-Jul	206	1.80	-	-	-
	26-Jul	208	2.20	-	-	-
	28-Jul	210	2.00	18.94	94	7.50
	31-Jul	213	2.10	-	-	-
	03-Aug	216	2.20	-	-	-
	04-Aug	217	2.20	21.39	99	7.68

Table A.1.3. Cont'd.

Limnocorral	Date	Day of year	Secchi (m)	Mean water column Temp. (C)	Cond. (µS/cm)	Dissolved oxygen (mg/L)
	10-Aug	223	2.30	-	-	-
	11-Aug	224	2.20	20.89	101	7.46
	13-Aug	226	2.00	-	-	-
	14-Aug	227	2.10	-	-	-
	19-Aug	232	1.80	-	-	-
	21-Aug	234	2.25	16.45	103	7.74
	22-Aug	235	-	17.35	103	8.07
	23-Aug	236	-	16.56	103	7.66
	25-Aug	238	1.80	18.11	105	8.96
OM83-6	23-Jun	175	1.00	16.17	83	8.66
	30-Jun	182	1.80	17.31	84	8.14
	05-Jul	187	1.80	15.79	86	8.16
	06-Jul	188	2.00	-	-	-
	07-Jul	189	2.00	-	-	-
	09-Jul	191	2.45	-	-	-
	10-Jul	192	2.40	-	-	-
	12-Jul	194	2.30	19.85	89	7.83
	14-Jul	196	2.80	-	-	-
	17-Jul	199	2.65	-	-	-
	20-Jul	202	2.90	19.54	89	7.97
	24-Jul	206	2.15	-	-	-
	26-Jul	208	1.95	-	-	-
	28-Jul	210	2.80	18.80	88	7.51
	31-Jul	213	2.60	-	-	-
	03-Aug	216	3.20	-	-	-
	04-Aug	217	3.30	21.34	91	7.52
	10-Aug	223	3.30	-	-	-
	11-Aug	224	3.30	20.90	91	7.62
	13-Aug	226	2.70	-	-	-
	14-Aug	227	3.00	-	-	-
	19-Aug	232	2.95	-	-	-
	21-Aug	234	3.60	16.53	90	8.00
	22-Aug	235	-	17.40	90	7.92
	23-Aug	236	-	16.60	89	7.61
	25-Aug	238	3.30	18.04	92	8.11
LM83-7	23-Jun	175	0.90	16.48	85	8.82
	30-Jun	182	1.60	17.21	72	8.04
	05-Jul	187	1.00	15.85	70	7.66
	06-Jul	188	1.70	-	-	-
	07-Jul	189	1.95	-	-	-
	09-Jul	191	2.10	-	-	-
	10-Jul	192	2.05	-	-	-
	12-Jul	194	2.00	19.63	74	7.23
	14-Jul	196	2.00	-	-	-
	17-Jul	199	2.20	-	-	-
	20-Jul	202	2.10	19.53	74	7.00
	24-Jul	206	2.05	-	-	-
	26-Jul	208	1.50	-	-	-
	28-Jul	210	1.25	18.66	77	6.23
	01-Aug	214	1.80	-	-	-
	03-Aug	216	1.95	-	-	-
	04-Aug	217	2.20	21.56	81	6.78
	10-Aug	223	1.90	-	-	-
	11-Aug	224	2.00	20.95	83	6.88
	13-Aug	226	1.75	-	-	-
	14-Aug	227	2.10	-	-	-
	19-Aug	232	1.80	-	-	-

Table A.1.3. Cont'd.

Limnocorral	Date	Day of year	Secchi (m)	Mean water column Temp. (C)	Cond. (µS/cm)	Dissolved oxygen (mg/L)
	21-Aug	234	1.80	16.38	82	7.94
	22-Aug	235	-	17.38	83	8.41
	23-Aug	236	-	16.60	82	7.76
	25-Aug	238	1.80	18.05	84	8.04
LMS83-8	23-Jun	175	1.00	16.34	84	8.53
	30-Jun	182	1.80	17.40	71	8.15
	05-Jul	187	0.60	15.82	69	7.95
	06-Jul	188	0.80	-	-	-
	07-Jul	189	1.00	-	-	-
	09-Jul	191	1.10	-	-	-
	10-Jul	192	1.20	-	-	-
	12-Jul	194	1.25	19.78	73	7.42
	14-Jul	196	1.05	-	-	-
	14-Jul	196	0.70	-	-	-
	17-Jul	199	1.20	-	-	-
	18-Jul	200	1.70	-	-	-
	18-Jul	200	0.50	-	-	-
	20-Jul	202	0.60	19.37	73	7.02
	20-Jul	202	1.05	-	-	-
	24-Jul	206	0.65	-	-	-
	24-Jul	206	1.30	-	-	-
	26-Jul	208	0.40	-	-	-
	26-Jul	208	1.10	-	-	-
	28-Jul	210	0.80	18.72	75	6.29
	31-Jul	213	0.30	-	-	-
	31-Jul	213	1.40	-	-	-
	03-Aug	216	0.95	-	-	-
	04-Aug	217	0.45	21.48	78	7.60
	04-Aug	217	1.10	-	-	-
	10-Aug	223	0.50	-	-	-
	10-Aug	223	1.20	-	-	-
	11-Aug	224	0.95	20.99	81	6.65
	13-Aug	226	1.30	-	-	-
	13-Aug	226	0.45	-	-	-
	14-Aug	227	0.90	-	-	-
	19-Aug	232	1.20	-	-	-
	19-Aug	232	0.40	-	-	-
	21-Aug	234	0.75	16.43	81	7.55
	22-Aug	235	0.30	-	-	-
	22-Aug	235	0.90	17.35	80	7.73
	23-Aug	236	-	16.68	80	7.31
	25-Aug	238	0.40	17.99	82	8.11
	25-Aug	238	0.95	-	-	-
L83-9	23-Jun	175	0.95	16.12	84	8.85
	30-Jun	182	1.60	17.76	85	8.54
	05-Jul	187	1.60	15.46	86	8.91
	06-Jul	188	1.10	-	-	-
	07-Jul	189	1.40	-	-	-
	09-Jul	191	1.10	-	-	-
	10-Jul	192	1.25	-	-	-
	12-Jul	194	1.20	19.19	89	8.06
	14-Jul	196	1.30	-	-	-
	17-Jul	199	1.40	-	-	-
	18-Jul	200	1.50	-	-	-
	20-Jul	202	1.70	19.95	89	8.12
	24-Jul	206	1.60	-	-	-
	26-Jul	208	1.80	-	-	-

Table A.1.3. Cont'd.

				Mean water column		
Limnocorral	Date	Day of year	Secchi (m)	Temp. (C)	Cond. (µS/cm)	Dissolved oxygen (mg/L)
	28-Jul	210	1.20	18.67	88	7.21
	31-Jul	213	1.60	-	-	-
	01-Aug	214	1.60	-	-	-
	03-Aug	216	1.80	-	-	-
	04-Aug	217	1.50	20.89	91	7.90
	10-Aug	223	1.65	-	-	-
	11-Aug	224	3.60	21.20	90	7.86
	13-Aug	226	1.40	-	-	-
	14-Aug	227	1.40	-	-	-
	19-Aug	232	1.45	-	-	-
	21-Aug	234	1.80	16.65	92	8.05
	22-Aug	235	-	16.97	-	-
	23-Aug	236	-	17.29	93	-
	25-Aug	238	1.85	17.94	94	8.00

Table A.1.4. Summary of physical conditions in the limnocorrals, 1984.

Limnocorral	Date	Day of year	Secchi (m)	Mean water column Temp. (C)	Cond. (µS/cm)	pH	Dissolved Oxygen (mg/L)
LMN84-1	29-Jun	181	1.80	15.17	75.8	6.3	7.7
	05-Jul	187	1.90	17.81	75.3	6.6	7.3
	13-Jul	195	2.10	19.79	78.5	6.7	6.9
	20-Jul	202	2.20	19.32	82.7	6.7	6.6
	26-Jul	208	1.90	19.42	88.4	6.8	6.4
	03-Aug	216	1.60	21.37	89.2	6.2	5.8
	12-Aug	225	2.20	19.74	94.8	6.8	7.0
	17-Aug	230	1.30	18.71	96.0	6.8	6.6
	29-Aug	242	2.60	16.80	90.1	-	7.6
	04-Sep	248	3.00	13.40	94.5	-	8.1
	30-Sep	274	3.20	5.20	91.2	7.2	10.7
CN84-2	29-Jun	181	1.85	18.26	92.7	7.5	8.5
	05-Jul	187	1.70	17.83	92.3	7.4	8.2
	13-Jul	195	2.10	17.75	94.6	7.6	8.6
	20-Jul	202	1.60	19.17	96.5	7.5	8.9
	26-Jul	208	1.60	19.55	97.6	7.5	9.7
	03-Aug	216	1.70	21.42	101.0	7.4	7.9
	12-Aug	225	2.50	19.80	108.0	7.4	7.6
	17-Aug	230	1.30	18.68	109.0	7.3	7.9
	29-Aug	242	1.20	16.65	103.9	-	8.9
	04-Sep	248	1.00	13.35	107.2	-	9.5
	30-Sep	274	1.20	5.10	104.0	7.5	10.6
S84-3	29-Jun	181	1.85	15.70	92.6	7.3	8.0
	05-Jul	187	1.80	17.83	92.8	6.8	7.2
	13-Jul	195	2.10	17.87	94.9	7.0	7.3
	20-Jul	202	1.80	19.30	97.4	6.7	7.1
	26-Jul	208	1.60	19.55	98.0	6.3	7.7
	03-Aug	216	1.70	21.51	102.3	6.4	7.8
	12-Aug	225	2.20	19.86	109.9	6.9	6.0
	17-Aug	230	2.20	18.79	111.0	6.8	6.4
	29-Aug	242	1.80	16.69	104.6	-	8.6
	04-Sep	248	2.60	13.30	109.5	-	9.1
	30-Sep	274	2.90	5.10	104.3	7.6	10.4
SD84-4	29-Jun	181	1.15	18.35	97.0	7.6	8.3
	05-Jul	187	1.20	17.97	100.0	8.5	9.0
	13-Jul	195	1.80	19.84	106.3	8.7	8.4
	20-Jul	202	0.80	19.32	106.0	8.5	9.9
	26-Jul	208	0.80	19.07	113.7	8.1	10.7
	03-Aug	216	1.00	21.55	120.9	8.8	7.2
	12-Aug	225	0.80	19.86	131.3	8.7	8.2
	17-Aug	230	0.60	19.12	132.9	8.9	8.1
	29-Aug	242	0.45	16.70	128.8	-	9.6
	04-Sep	248	0.20	13.32	134.7	-	9.0
	30-Sep	274	0.80	5.20	129.0	8.6	11.0
LM84-5	29-Jun	181	1.90	18.00	74.1	6.2	7.2
	05-Jul	187	2.20	17.77	73.0	6.2	6.4
	13-Jul	195	2.20	19.88	83.4	6.5	6.7
	20-Jul	202	1.80	19.63	80.5	6.4	5.8
	26-Jul	208	1.80	19.84	82.6	5.8	5.8
	03-Aug	216	0.90	21.62	87.2	7.1	6.5
	12-Aug	225	1.25	19.84	92.7	6.8	5.6
	17-Aug	230	1.40	18.91	93.5	6.5	5.4
	29-Aug	242	1.42	16.65	87.9	-	6.7
	04-Sep	248	1.80	13.39	92.8	-	9.6
	30-Sep	274	2.40	5.10	89.0	7.1	9.6

Table A.1.4. Cont'd.

				Mean water column			
Limnocorral	Date	Day of year	Secchi (m)	Temp. (C)	Cond. (µS/cm)	pH	Dissolved Oxygen (mg/L)
Hg84-6	29-Jun	181	2.10	18.50	93.5	7.5	8.1
	05-Jul	187	2.20	17.93	93.0	7.3	7.9
	13-Jul	195	2.20	19.74	95.1	7.6	8.3
	20-Jul	202	2.40	19.48	96.5	7.2	8.5
	26-Jul	208	2.00	19.68	97.0	7.4	8.9
	03-Aug	216	2.80	21.48	100.2	7.5	7.7
	12-Aug	225	2.70	20.03	105.5	7.6	7.7
	17-Aug	230	2.40	18.91	108.4	7.5	8.0
	29-Aug	242	2.00	16.60	102.7	-	9.2
	04-Sep	248	1.90	13.28	108.0	-	9.7
	30-Sep	274	2.45	5.10	103.1	7.7	10.6
C84-7	29-Jun	181	2.00	18.59	93.0	7.3	8.1
	05-Jul	187	2.05	18.03	93.0	7.3	7.9
	13-Jul	195	1.90	19.82	94.9	7.5	8.5
	20-Jul	202	2.00	19.46	96.9	7.0	8.9
	26-Jul	208	1.90	20.04	97.0	5.8	9.3
	03-Aug	216	2.30	21.58	100.0	7.4	7.8
	12-Aug	225	2.20	20.03	105.7	7.4	6.7
	17-Aug	230	2.00	18.87	108.9	7.2	7.6
	29-Aug	242	1.80	16.70	102.0	-	8.8
	04-Sep	248	1.60	13.33	106.6	-	9.6
	30-Sep	274	2.30	5.10	102.0	7.4	10.4
L84-8	29-Jun	181	1.40	17.00	92.0	7.4	8.6
	05-Jul	187	1.25	18.09	91.0	7.3	7.9
	20-Jul	202	1.20	19.16	93.4	7.4	8.8
	26-Jul	208	1.60	19.82	94.0	7.4	9.3
	03-Aug	216	1.90	21.85	98.0	6.9	5.0
	12-Aug	225	1.20	20.06	104.6	7.5	7.5
	17-Aug	230	1.20	18.97	105.7	7.3	7.7
	29-Aug	242	1.00	16.90	102.4	-	8.7
	04-Sep	248	1.00	13.77	107.0	-	9.3
	30-Sep	274	1.00	5.37	106.6	7.4	10.2

Appendix 2

Activity of radioisotopes in various fractions of water columns of limnocorrals, 1981-1984

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Table A.2.1. ^{203}Hg activity (dpm/L)in the water column; 1981 limnocorral experiment.

Limnocorral	Fraction	Date (day of year)				
		July 3 (184)	July 7 (188)	July 19 (200)	Aug 2 (214)	Aug 23 (235)
C81-1	Charcoal	6279	3919	1595	486	651
	Anion	5260	3160	998	881	205
	Cation	844	526	584	628	326
	Sulfide	3330	1460	584	628	326
	GFC	14544	10726	3867	2081	1156
	MP	3321	2333	1926	1170	421
	Netplankton	95	98	37	28	16
	Total activity	33674	22221	9591	5902	3100
	Dissolved (%)	47	41	39	44	49
	Particulate (%)	53	59	61	56	51
	% of time 0	100	66	28	18	9
LM81-2	Charcoal	6623	2709	1516	1563	695
	Anion	4833	3528	2126	1377	1349
	Cation	933	605	307	147	233
	Sulfide	3979	2944	1516	840	298
	GFC	14574	8609	2686	1326	1409
	MP	3133	2616	1830	4130	528
	Netplankton	72	70	56	67	174
	Total activity	34147	21081	10037	9449	4686
	Dissolved (%)	48	46	54	42	55
	Particulate (%)	52	54	46	58	45
	% of time 0	100	63	30	28	14
PN81-3	Charcoal	8700	5826	2209	1042	740
	Anion	3000	3195	1272	395	449
	Cation	372	316	63	-	-
	Sulfide	951	391	258	1956	314
	GFC	10158	10842	8812	5195	2730
	MP	2470	1628	1195	1088	3237
	Netplankton	100	207	123	91	100
	Total activity	25751	22405	13933	9767	7570
	Dissolved (%)	51	43	27	35	20
	Particulate (%)	49	57	73	65	80
	% of time 0	100	67	41	29	22
LS81-4	Charcoal	5365	4914	2130	809	237
	Anion	6530	3456	1170	616	30
	Cation	179	384	209	-	-
	Sulfide	2735	409	140	102	142
	GFC	8737	9798	4070	3847	1233
	MP	4174	1905	958	1053	1205
	Netplankton	60	112	56	40	37

Table A.2.1. Cont'd.

Limnocorral	Fraction	Date (day of year)				
		July 3 (184)	July 7 (188)	July 19 (200)	Aug 2 (214)	Aug 23 (235)
	Total activity	27781	20977	8733	6467	2884
	Dissolved (%)	53	44	42	24	14
	Particulate (%)	47	56	58	76	86
	% of time 0	100	62	26	19	9
C81-5	Charcoal	9028	6047	2426	2016	1612
	Anion	4170	2088	1237	233	619
	Cation	1105	409	133	151	123
	Sulfide	2884	2542	253	140	516
	GFC	23453	9267	6621	5198	1274
	MP	2844	1651	588	1137	1370
	Netplankton	63	123	98	121	79
	Total activity	43547	22128	11356	8995	5593
	Dissolved (%)	39	50	36	28	51
	Particulate (%)	61	50	64	72	49
	% of time 0	100	66	34	27	17
HS81-6	Charcoal	10005	4240	2253	1726	895
	Anion	7705	3453	1337	381	286
	Cation	1033	651	170	0	119
	Sulfide	1377	1107	340	193	119
	GFC	8674	10058	4577	2842	1188
	MP	7184	4081	1784	2070	933
	Netplankton	58	79	40	42	28
	Total activity	35977	23591	10460	7212	3540
	Dissolved (%)	56	40	39	32	40
	Particulate (%)	44	60	61	68	60
	% of time 0	100	70	31	21	11
C81-7	Charcoal	11386	5435	2772	1293	1091
	Anion	6184	4244	1291	700	1147
	Cation	672	228	174	372	58
	Sulfide	547	484	207	326	279
	GFC	8847	9253	6116	4942	1716
	MP	2777	1349	595	819	2016
	Netplankton	126	170	193	67	237
	Total activity	30412	20993	11156	8451	6307
	Dissolved (%)	62	49	40	32	41
	Particulate (%)	38	51	60	68	59
	% of time 0	100	62	33	25	19

Table A.2.2. ^{203}Hg in the water column; 1982 limnocorral experiment.

Limnocorral	Fraction	Date (Day of Year)																					
		July 4 (185)	July 5 (186)	July 6 (187)	July 7 (188)	July 8 (189)	July 9 (190)	July 12 (193)	July 14 (195)	July 15 (196)	July 19 (200)	July 21 (202)	July 26 (207)	July 29 (210)	Aug 2 (214)	Aug 6 (218)	Aug 9 (221)	Aug 13 (225)	Aug 16 (228)	Aug 20 (232)	Aug 23 (235)	Aug 26 (238)	Aug 30 (242)
FM82-1	Total (dpm/mL)	65.0	61.4	59.6	55.6	54.6	50.0	37.4	34.6	30.6	20.9	19.4	17.2	14.4	12.0	10.8	10.8	10.7	8.5	5.9	6.2	5.7	4.5
	Dissolved (%)	70	65	65	59	63	61	58	61	65	65	69	63	54	55	61	50	55	56	36	41	52	53
	Particulate (%)	30	35	35	41	37	39	42	39	35	35	31	37	46	45	39	50	45	44	64	59	48	47
	% of time 0	100	94	92	86	84	77	58	53	47	32	30	26	22	18	17	17	16	13	9	10	9	7
LM82-3	Total (dpm/mL)	37.3	37.2	31.6	31.4	29.9	29.1	20.6	20.4	18.9	15.7	14.1	13.0	10.5	8.0	5.2	5.1	5.7	4.0	2.0	2.3	1.8	1.4
	Dissolved (%)	76	39	75	62	65	66	61	65	68	70	69	79	74	58	70	61	68	63	22	17	44	40
	Particulate (%)	24	61	25	38	35	34	39	35	32	30	31	21	26	42	30	39	32	37	78	83	56	60
	% of time 0	100	100	85	84	80	78	55	55	51	42	38	35	28	21	14	14	15	11	5	6	5	4
HM82-4	Total (dpm/mL)	49.6	45.9	40.6	38.9	39.5	38.5	26.9	25.6	24.0	19.9	18.5	14.7	11.9	8.8	5.2	5.0	5.0	3.7	1.2	2.1	2.3	1.9
	Dissolved (%)	74	64	69	65	67	67	80	65	67	56	54	62	51	49	56	39	48	50	0	36	44	44
	Particulate (%)	26	36	31	35	33	33	20	35	33	44	46	38	49	51	44	61	52	50	100	64	56	56
	% of time 0	100	93	82	79	80	78	54	52	48	40	37	30	24	18	10	10	10	7	2	4	5	4
OC82-5	Total (dpm/mL)	39.6	38.5	32.8	32.7	31.2	29.6	20.1	18.7	17.5	14.0	11.6	8.9	6.6	6.6	5.2	4.3	5.7	4.8	2.7	3.4	2.7	3.1
	Dissolved (%)	70	65	67	61	62	62	63	61	63	60	56		59	57	61	59	61	79	62	60	57	40
	Particulate (%)	30	35	33	39	38	38	37	39	37	40	44	100	41	43	39	41	39	21	38	40	43	60
	% of time 0	100	97	83	83	79	75	51	47	44	35	29	22	17	17	13	11	14	12	7	9	5	8
C82-6	Total (dpm/mL)	44.5	43.4	38.6	40.4	38.3	35.1	27.6	26.8	30.0	21.9	18.6	17.5	14.7	13.5	10.3	8.5	8.3	6.0	4.3	4.7	3.9	3.3
	Dissolved (%)	70	70	66	69	70	69	69	71	70	69	69	70	68	65	61	67	61	56	50	55	59	67
	Particulate (%)	30	30	34	31	30	31	31	29	30	31	31	30	32	35	39	33	39	44	50	45	41	33
	% of time 0	100	98	87	91	86	79	62	60	67	49	42	39	33	30	23	19	19	13	10	11	9	7
HS82-7	Total (dpm/mL)	46.3	43.8	39.9	41.8	37.8	36.0	25.4	25.8	24.0	17.1	14.9	14.4	10.7	10.3	7.2	5.8	5.7	3.8	3.1	3.0	2.9	2.4
	Dissolved (%)	68	60	66	58	56	57	47	52	53	62	53	42	46	47	52	51	48	50	40	55	55	50
	Particulate (%)	32	40	34	42	44	43	53	48	47	38	47	58	54	53	48	49	52	50	60	45	45	50
	% of time 0	100	95	86	90	82	78	55	56	52	37	32	31	23	22	16	13	12	8	7	6	6	5
OM82-8	Total (dpm/mL)	41.6	38.4	35.4	31.8	31.4	29.4	21.9	20.1	18.7	16.4	13.3	12.7	11.3	11.7	9.1	8.5	8.6	8.1	7.1	7.0	5.7	5.2
	Dissolved (%)	72	68	60	60	57	58	56	59	59	58	58	59	61	47	53	43	46	43	48	44	50	48
	Particulate (%)	28	32	40	40	43	42	44	41	41	42	42	41	39	53	47	57	54	57	52	56	50	52
	% of time 0	100	92	85	76	75	71	53	48	45	39	32	31	27	28	22	20	21	19	17	17	14	13

Table A.2.3. ^{203}Hg in the water column; 1983 limnocorral experiment.

Limnocorral	Fraction	Date (Day of Year)																		
		July 7 (188)	July 8 (189)	July 10 (191)	July 11 (192)	July 13 (194)	July 15 (196)	July 18 (199)	July 19 (200)	July 21 (202)	July 25 (206)	July 29 (210)	Aug 1 (213)	Aug 4 (216)	Aug 8 (220)	Aug 11 (223)	Aug 15 (227)	Aug 20 (232)	Aug 24 (236)	Aug 29 (241)
DFM83-1	Total (dpm/mL)	86.2	70.9	71.4	67.8	59.6	48.1	-	33.3	31.0	28.9	22.9	21.1	19.5	17.4	12.6	13.4	10.0	9.3	7.8
	Dissolved (%)	67	57	62	61	58	62	-	68	65	68	53	57	54	64	60	62	60	67	70
	Particulate (%)	33	43	38	39	42	38	-	32	35	32	47	43	46	36	40	38	40	33	30
	% of time 0	100	82	83	78	69	56	-	42	37	34	27	24	22	20	14	16	12	11	9
SP83-2	Total (dpm/mL)	63.2	49.8	49.2	49.2	46.0	43.0	-	31.4	29.9	28.0	24.8	24.1	20.5	20.5	16.1	14.6	11.2	9.1	7.3
	Dissolved (%)	65	60	37	36	40	31	-	32	35	50	44	38	35	29	36	29	31	35	51
	Particulate (%)	35	40	63	64	60	69	-	68	65	50	56	62	65	71	64	71	69	65	49
	% of time 0	100	79	77	77	73	67	-	53	51	44	39	38	32	33	26	23	18	14	12
LM83-3	Total (dpm/mL)	61.9	51.6	51.2	49.9	44.5	40.3	33.1	-	29.3	27.4	22.4	19.8	16.8	21.7	9.2	8.7	5.8	5.5	5.4
	Dissolved (%)	78	74	74	75	75	70	68	-	70	70	61	60	57	60	53	55	58	72	73
	Particulate (%)	22	26	26	25	25	30	32	-	30	30	39	40	43	40	47	45	42	28	27
	% of time 0	100	83	82	79	70	64	52	-	48	43	35	32	27	34	14	14	9	9	9
C83-4	Total (dpm/mL)	59.8	53.3	47.8	46.9	43.2	40.2	31.6	-	26.8	23.7	17.9	14.3	11.8	10.8	7.7	8.9	6.0	5.1	4.0
	Dissolved (%)	73	68	64	64	61	58	60	-	58	57	57	67	56	66	58	76	67	84	76
	Particulate (%)	27	32	36	36	39	42	40	-	42	43	43	33	44	34	42	24	33	16	24
	% of time 0	100	87	78	77	71	67	52	-	46	39	30	24	19	18	13	15	10	9	7
OAS83-5	Total (dpm/mL)	60.2	61.3	51.0	48.7	41.0	36.3	-	24.6	22.7	17.9	15.6	14.9	12.0	10.4	7.8	8.1	7.4	6.6	6.5
	Dissolved (%)	73	66	66	64	63	59	-	61	61	59	54	51	56	57	57	63	67	70	63
	Particulate (%)	27	34	34	36	37	41	-	39	39	41	46	49	44	43	43	37	33	30	37
	% of time 0	100	100	83	79	67	59	-	44	35	29	26	24	20	18	13	13	12	11	11
OM83-6	Total (dpm/mL)	79.2	57.5	49.3	46.0	41.1	35.1	30.9	-	25.8	22.7	17.9	17.5	11.6	13.2	11.4	9.9	7.4	7.3	6.7
	Dissolved (%)	67	65	68	66	66	69	71	-	75	69	59	67	62	70	69	75	76	80	82
	Particulate (%)	33	35	32	34	34	31	29	-	25	31	41	33	38	30	31	25	24	20	18
	% of time 0		100	85	81	72	61	54	-	46	39	31	30	20	23	20	17	13	13	12
LM83-7	Total (dpm/mL)	68.5	58.9	53.6	52.5	46.1	40.9	33.7	-	29.8	27.4	22.7	20.1	13.6	12.1	9.4	8.7	6.4	5.5	4.8
	Dissolved (%)	76	74	75	74	73	70	70	-	71	74	54	56	52	55	55	75	60	66	65
	Particulate (%)	24	26	25	26	27	30	30	-	29	26	46	44	48	45	45	25	40	34	35
	% of time 0	100	86	79	76	67	60	49	-	49	40	33	29	20	18	14	13	9	8	7
LMS83-8	Total (dpm/mL)	53.0	52.3	48.0	46.4	41.6	35.9	25.4	-	20.0	15.2	9.9	7.7	6.8	4.9	5.0	7.4	3.2	4.0	2.1
	Dissolved (%)	75	68	68	67	72	59	63	-	69	71	69	65	61	73	68	71	81	78	75
	Particulate (%)	25	32	32	33	28	41	37	-	31	29	31	35	39	27	32	29	19	22	25
	% of time 0	100	95	86	84	75	65	46	-	37	28	18	14	12	9	9	14	6	8	4

Table A.2.4. ^{203}Hg in the water column; 1984 limnocorral experiment.

Limnocorral	Fraction	Date (day of year)																				
		June 27 (179)	June 29 (181)	July 2 (184)	July 5 (187)	July 9 (191)	July 13 (195)	July 16 (198)	July 19 (201)	July 23 (205)	July 26 (208)	July 31 (213)	Aug 3 (216)	Aug 6 (219)	Aug 12 (225)	Aug 15 (228)	Aug 17 (230)	Aug 21 (234)	Aug 26 (239)	Aug 29 (242)	Sept 4 (248)	Sept 30 (274)
SP84-3	Total (dpm/mL)	26.7	24.5	26.9	15.9	14.2	11.7	11.5	7.9	8.0	8.9	5.0	3.5	9.3	3.3	3.2	3.3	2.9	2.3	2.0	1.9	0.0
	Dissolved (%)	-	-	73	68	68	61	60	58	41	56	60	34	29	60	55	60	58	4	76	69	-
	Particulate (%)	-	-	27	32	32	39	40	42	59	44	40	66	71	40	45	40	42	96	24	31	-
	% of time 0	100	92	101	60	53	44	43	30	30	33	19	13	35	12	12	12	11	9	7	7	0
SD84-4	Total (dpm/mL)	20.1	15.9	21.9	10.8	10.5	9.4	12.8	6.6	5.9	5.5	2.4	2.6	5.9	2.8	5.2	2.9	4.5	2.9	3.4	2.8	0.8
	% of time 0	100	79	109	54	52	47	64	33	30	27	12	13	29	14	26	14	22	14	17	14	4
LM84-5	Total (dpm/mL)	39.9	20.0	24.5	11.2	9.4	8.0	10.0	6.4	4.5	3.3	6.3	2.6	3.4	0.9	0.9	1.0	0.9	2.4	0.8	2.2	0.0
	% of time 0	-	100	100	50	42	36	45	29	20	15	28	12	15	4	4	4	4	11	4	10	0
CHg84-6	Total (dpm/mL)	23.4	20.0	23.9	11.7	10.4	8.6	11.6	7.4	5.9	5.6	4.0	4.3	5.5	2.6	2.2	2.8	2.5	2.8	2.7	2.3	0.6
	% of time 0	100	85	102	50	44	37	50	32	25	24	17	18	24	11	9	12	11	12	12	10	3
C84-7	Total (dpm/mL)	24.3	19.3	23.4	11.2	12.2	7.9	10.8	5.9	5.1	5.6	2.6	2.4	5.9	3.0	1.4	3.2	2.5	2.4	1.6	2.0	0.3
	Dissolved (%)	-	-	59	63	57	63	67	68	64	68	66	60	53	50	52	50	55	48	80	50	-
	Particulate (%)	-	-	41	37	43	37	33	32	36	32	34	40	47	50	48	50	45	52	20	50	-
	% of time 0	100	79	96	46	50	33	44	24	21	23	11	10	24	12	6	13	10	10	7	8	1

Table A.2.5. Tritium in the water column (dpm/mL); 1981 limnocorral experiment.

Date	Day of year	Limnocorral						
		C81-1	LM81-2	PN81-3	LS81-4	C81-5	HS81-6	C81-7
June 21	172	1123	1117	1165	942	989	1001	992
June 23	174	1256	-	1155	-	1019	-	1019
June 24	175	-	1012	-	880	-	874	-
July 1	182	1083	1040	976	896	906	933	925
July 7	188	1145	1084	1062	1025	1026	1067	1053
July 10	191	1328	1225	1244	1161	1169	1226	1179
July 19	200	1089	996	1046	970	961	1016	1004
July 24	205	1056	1000	991	880	908	911	935
July 28	209	1002	969	965	856	880	894	907
Aug 3	215	998	927	979	825	921	932	958
Aug 15	227	991	963	974	883	903	893	919
Aug 24	236	868	842	845	769	768	744	809

Table A.2.6. Tritium in the water column (dpm/mL); 1982 limnocorral experiment.

Date	Day of year	Limnocorral							
		FM82-1	SC82-2	LM82-3	HM82-4	OC82-5	C82-6	HS82-7	OM82-8
July 4	185	1221	-	709	935	758	856	879	778
July 5	186	1218	-	731	988	785	911	918	782
July 6	187	1284	-	741	980	779	889	926	801
July 7	188	1276	965	796	1063	789	909	929	795
July 8	189	1174	925	739	966	748	907	915	951
July 9	190	1141	-	720	910	687	846	851	761
July 12	193	1153	916	719	907	648	832	881	753
July 14	195	1111	910	721	898	632	824	844	740
July 15	196	1121	897	691	899	619	826	856	722
July 19	200	1042	896	704	904	603	831	866	752
July 21	202	1048	912	738	914	605	849	878	718
July 26	207	984	854	669	849	535	801	846	726
July 29	210	1131	913	707	884	518	836	835	711
Aug 2	214	935	869	673	862	518	864	895	739
Aug 6	218	870	716	549	714	369	653	677	582
Aug 9	221	848	829	633	837	437	826	846	723
Aug 13	225	843	846	663	827	426	808	804	671
Aug 16	228	829	817	668	804	1018	736	783	639
Aug 20	232	824	830	629	816	388	765	776	676
Aug 23	235	868	860	650	823	405	793	819	693
Aug 26	238	863	846	662	870	374	805	832	716
Aug 30	242	885	864	679	834	366	829	813	674

Table A.2.7. Tritium in the water column (dpm/mL); 1983 limnocorral experiment.

Date	Day of year	Limnocorral							
		DFM83-1	SP83-2	LM83-3	C83-4	OAS83-5	OM83-6	LM83-7	LMS83-8
July 7	188	1198	867	921	1058	924	1324	1003	818
July 8	189	1066	756	766	820	885	896	901	752
July 10	191	1145	812	789	824	891	896	885	775
July 11	192	1188	815	801	833	887	882	904	751
July 13	194	1179	838	841	855	921	864	893	782
July 15	196	1237	835	838	889	933	833	899	768
July 18	199	-	-	773	779	-	781	851	746
July 19	200	-	-	-	-	935	-	-	-
July 21	202	1214	859	842	889	940	861	935	833
July 25	206	1196	844	851	877	923	796	894	784
July 29	210	1056	761	793	820	846	763	811	734
Aug 1	213	1072	725	800	782	826	743	791	712
Aug 4	216	1065	657	778	756	844	719	774	703
Aug 8	220	1009	660	748	774	815	713	776	724
Aug 11	223	922	602	707	704	761	689	739	606
Aug 15	227	1001	664	747	776	728	696	758	678
Aug 20	232	892	611	689	708	721	652	707	613
Aug 24	236	871	572	702	718	704	653	678	645
Aug 29	241	886	572	645	654	675	612	651	617

Table A.2.8. Tritium in the water column (dpm/mL); 1984 limnocorral experiment.

Date	Day of year	Limnocorral						
		LMN84-1	CN84-2	SP84-3	SD84-4	LM84-5	CHg84-6	C84-7
June 27	179	44	48	36	29	44	35	35
June 29	181	54	53	70	61	71	67	69
July 2	184	63	64	62	56	69	58	69
July 5	187	74	67	59	54	63	59	64
July 9	191	56	66	67	56	68	57	58
July 13	195	68	58	50	56	66	58	57
July 16	198	64	63	61	67	72	64	66
July 19	201	56	57	59	54	67	62	63
July 23	205	75	63	65	52	65	61	60
July 26	208	60	50	29	48	61	57	53
July 31	213	45	43	46	46	49	43	43
Aug 3	216	36	26	31	24	30	36	31
Aug 6	219	136	77	72	65	76	70	71
Aug 12	225	77	40	48	43	49	38	29
Aug 15	228	58	46	54	41	53	51	67
Aug 17	230	56	40	49	46	49	42	45
Aug 21	234	34	31	51	48	63	58	62
Aug 26	239	65	38	46	49	60	54	57
Aug 29	242	56	31	51	54	71	61	63
Sept 4	248	60	27	52	50	63	57	63
Sept 30	274	51	16	50	40	53	57	63

Table A.2.9. ^{75}Se in the water column (dpm/L); 1981 limnocorral experiment.

Limnocorral	Fraction	Date (day of year)				
		July 3 (184)	July 7 (188)	July 19 (200)	Aug 2 (214)	Aug 23 (235)
C81-1	Charcoal	2033	2164	2233	2495	3978
	Anion	1597	2248	2163	1835	2263
	Cation	53	141	114	135	159
	Sulfide	19	16	50	57	112
	GFC	5030	2240	676	822	710
	MP	958	561	275	371	279
	Netplankton	52	86	33	17	20
	Total activity	9742	7457	5545	5731	7522
	Dissolved (%)	38	61	82	79	87
	Particulate (%)	62	39	18	21	13
	% of time 0	100	77	57	59	77
LM81-2	Charcoal	1521	2143	2156	1681	1660
	Anion	1488	1911	1759	2188	2195
	Cation	221	119	126	191	45
	Sulfide	64	72	14	0	83
	GFC	5148	1458	553	469	458
	MP	929	410	889	406	170
	Netplankton	36	54	50	39	51
	Total activity	9408	6166	5547	4974	4662
	Dissolved (%)	35	69	73	82	85
	Particulate (%)	65	31	27	18	15
	% of time 0	100	63	57	51	48
PN81-3	Charcoal	1652	3144	4122	3128	3247
	Anion	1397	2322	1891	2131	2266
	Cation	0	90	0	0	61
	Sulfide	178	0	0	190	130
	GFC	3303	1678	944	1139	1051
	MP	1413	506	216	131	931
	Netplankton	59	88	56	65	65
	Total activity	8002	7828	7228	6785	7751
	Dissolved (%)	40	71	83	80	74
	Particulate (%)	60	29	17	20	26
	% of time 0	100	80	74	70	80
LS81-4	Charcoal	841	2163	3431	3061	3019
	Anion	1791	2088	1972	1725	2138
	Cation	95	0	0	0	0
	Sulfide	38	0	0	103	125
	GFC	2966	1953	713	1272	738
	MP	1866	766	500	491	394
	Netplankton	22	59	22	50	81

Table A.2.9. Cont'd.

Limnocorral	Fraction	Date (day of year)				
		July 3 (184)	July 7 (188)	July 19 (200)	Aug 2 (214)	Aug 23 (235)
	Total activity	7617	7028	6638	6702	6494
	Dissolved (%)	36	60	81	73	81
	Particulate (%)	64	40	19	27	19
	% of time 0	100	72	68	69	67
C81-5	Charcoal	1303	2803	3775	5009	3263
	Anion	2028	1806	2347	681	2600
	Cation	138	0	0	106	166
	Sulfide	138	0	0	88	122
	GFC	80794	1844	794	1178	397
	MP	1628	644	188	278	450
	Netplankton	21	61	33	109	0
	Total activity	86049	7158	7137	7449	6997
	Dissolved (%)	4	64	86	79	88
	Particulate (%)	96	36	14	21	12
	% of time 0	100	73	73	76	72
HS81-6	Charcoal	1769	2238	4116	3944	3363
	Anion	2247	2003	1931	769	1431
	Cation	150	75	109	0	0
	Sulfide	0	0	0	103	191
	GFC	2831	2247	959	928	600
	MP	2691	1228	813	447	450
	Netplankton	21	64	23	77	36
	Total activity	9688	7791	7928	6191	6034
	Dissolved (%)	43	55	78	78	83
	Particulate (%)	57	45	22	22	17
	% of time 0	100	80	81	64	62
C81-7	Charcoal	1631	2169	4603	4013	2850
	Anion	1444	2544	1913	1244	3144
	Cation	0	103	0	0	416
	Sulfide	0	0	184	0	116
	GFC	3044	2178	769	897	638
	MP	1641	572	178	338	497
	Netplankton	46	100	71	45	109
	Total activity	7759	7566	7647	6491	7659
	Dissolved (%)	40	64	88	81	85
	Particulate (%)	60	36	12	19	15
	% of time 0	100	78	78	67	79

Table A.2.10. ^{134}Cs in the water column (dpm/L); 1981 limnocorral experiment.

Limnocorral	Fraction	Date (day of year)				
		July 3 (184)	July 7 (188)	July 19 (200)	Aug 2 (214)	Aug 23 (235)
C81-1	Charcoal	1091	1228	1014	915	647
	Anion	19	0	0	0	0
	Cation	6609	5014	1544	900	350
	Sulfide	0	0	0	0	0
	GFC	2902	2804	904	790	373
	MP	277	435	307	244	85
	Netplankton	13	15	7	10	4
	Total activity	10911	9495	3777	2859	1460
	Dissolved (%)	71	66	68	63	68
	Particulate (%)	29	34	32	37	32
	% of time 0	100	87	35	26	13
LM81-2	Charcoal	1251	1029	416	154	179
	Anion	0	25	0	0	0
	Cation	11736	8023	1592	1212	1064
	Sulfide	0	0	28	0	0
	GFC	2169	1544	408	228	261
	MP	104	211	398	212	42
	Netplankton	9	10	4	7	15
	Total activity	15269	10843	2845	1813	1562
	Dissolved (%)	85	84	72	75	80
	Particulate (%)	15	16	28	25	20
	% of time 0	100	71	19	12	10
PN81-3	Charcoal	1288	1408	1359	603	508
	Anion	0	0	0	0	0
	Cation	6106	5977	2634	2889	2764
	Sulfide	0	0	0	0	0
	GFC	1579	1576	831	805	778
	MP	204	165	63	116	292
	Netplankton	8	17	8	19	23
	Total activity	9186	9143	4895	4431	4364
	Dissolved (%)	81	81	82	79	75
	Particulate (%)	19	19	18	21	25
	% of time 0	100	100	53	48	48
LS81-4	Charcoal	419	662	213	0	264
	Anion	0	0	0	0	0
	Cation	4490	3553	722	444	259
	Sulfide	87	815	282	208	0
	GFC	4731	3736	815	861	801
	MP	1157	468	389	500	625
	Netplankton	9	14	5	9	5

Table A.2.10. Cont'd.

Limnocorral	Fraction	Date (day of year)				
		July 3 (184)	July 7 (188)	July 19 (200)	Aug 2 (214)	Aug 23 (235)
	Total activity	10894	9247	2426	2023	1954
	Dissolved (%)	46	54	50	32	27
	Particulate (%)	54	46	50	68	73
	% of time 0	100	85	22	19	18
C81-5	Charcoal	505	1000	505	1185	435
	Anion	0	0	0	0	0
	Cation	5625	4588	2236	1727	2319
	Sulfide	0	0	0	0	0
	GFC	1190	1264	356	690	713
	MP	148	120	134	88	380
	Netplankton	5	10	9	17	16
	Total activity	7473	6982	3240	3706	3863
	Dissolved (%)	82	80	85	79	71
	Particulate (%)	18	20	15	21	29
	% of time 0	100	93	43	50	52
HS81-6	Charcoal	398	204	134	0	0
	Anion	0	111	130	0	0
	Cation	3116	2171	569	384	310
	Sulfide	88	88	0	0	0
	GFC	6792	4718	1421	884	431
	MP	2880	2120	907	625	375
	Netplankton	15	160	5	6	14
	Total activity	13288	9572	3167	1900	1130
	Dissolved (%)	27	27	26	20	27
	Particulate (%)	73	73	74	80	73
	% of time 0	100	72	24	14	9
C81-7	Charcoal	903	1074	1426	537	505
	Anion	0	0	0	0	0
	Cation	7273	6759	2833	3190	3370
	Sulfide	0	0	0	0	0
	GFC	1546	1296	583	838	810
	MP	181	134	116	130	366
	Netplankton	12	12	51	19	296
	Total activity	9915	9275	5010	4713	5347
	Dissolved (%)	82	84	85	79	72
	Particulate (%)	18	16	15	21	28
	% of time 0	100	94	51	48	54

Table A.2.11. ^{60}Co in the water column (dpm/L); 1981 limnocorral experiment.

Limnocorral	Fraction	Date (day of year)				
		July 3 (184)	July 7 (188)	July 19 (200)	Aug 2 (214)	Aug 23 (235)
C81-1	Charcoal	4473	5114	3681	3874	4504
	Anion	1747	2702	1953	1215	1458
	Cation	1598	1113	442	293	477
	Sulfide	104	0	0	285	0
	GFC	70935	35465	5631	3098	1259
	MP	854	1156	506	1170	314
	Netplankton	170	147	30	46	32
	Total activity	79881	45698	12243	9981	8044
	Dissolved (%)	10	20	50	57	80
	Particulate (%)	90	80	50	43	20
	% of time 0	100	57	15	12	10
LM81-2	Charcoal	1102	1368	1269	1181	1020
	Anion	328	786	700	768	1092
	Cation	404	308	354	171	304
	Sulfide	0	0	0	0	0
	GFC	74190	26434	5592	1591	2544
	MP	351	438	5276	300	240
	Netplankton	214	104	29	40	135
	Total activity	76589	29438	13220	4052	5335
	Dissolved (%)	2	8	18	52	45
	Particulate (%)	98	92	82	48	55
	% of time 0	100	38	17	5	7
PN81-3	Charcoal	17838	19627	27825	22610	26058
	Anion	11557	16722	13381	12629	16103
	Cation	5696	2835	1825	2381	4000
	Sulfide	0	0	0	487	0
	GFC	23124	21711	9412	5299	5175
	MP	1485	1093	948	794	4216
	Netplankton	209	326	85	108	109
	Total activity	59908	62313	53476	44308	55662
	Dissolved (%)	59	63	80	86	83
	Particulate (%)	41	37	20	14	17
	% of time 0	100	104	89	74	93
LS81-4	Charcoal	8160	10763	9969	7021	6041
	Anion	6680	7918	5309	3247	3629
	Cation	11062	2454	948	629	1093
	Sulfide	0	0	0	0	124
	GFC	32804	29784	10918	11278	6024
	MP	1629	1505	1258	2010	1237
	Netplankton	98	196	72	104	72

Table A.2.11. Cont'd.

Limnocorral	Fraction	Date (day of year)				
		July 3 (184)	July 7 (188)	July 19 (200)	Aug 2 (214)	Aug 23 (235)
	Total activity	60433	52619	28474	24290	18220
	Dissolved (%)	43	40	57	45	60
	Particulate (%)	57	60	43	55	40
	% of time 0	100	87	47	40	30
C81-5	Charcoal	6619	13165	16691	21670	11866
	Anion	4485	5412	8041	742	7237
	Cation	7062	1794	2113	299	2711
	Sulfide	0	0	0	0	124
	GFC	45010	32113	12814	12041	7928
	MP	1082	1495	753	2052	2258
	Netplankton	101	292	179	178	89
	Total activity	64359	54271	40592	36982	32212
	Dissolved (%)	28	38	66	61	68
	Particulate (%)	72	62	34	39	32
	% of time 0	100	84	63	57	50
HS81-6	Charcoal	10753	6928	6289	4701	1155
	Anion	4113	4567	2732	691	1134
	Cation	7505	2216	753	0	351
	Sulfide	0	0	0	0	0
	GFC	43794	31485	9268	5454	2485
	MP	3722	2278	1041	1021	546
	Netplankton	115	140	35	58	56
	Total activity	70002	47614	20118	11924	5726
	Dissolved (%)	32	29	49	45	46
	Particulate (%)	68	71	51	55	54
	% of time 0	100	68	29	17	8
C81-7	Charcoal	13072	11144	22103	18856	11103
	Anion	3392	9299	6969	3464	10216
	Cation	3680	7000	1072	402	2505
	Sulfide	0	0	0	0	144
	GFC	38959	32619	11619	15082	7680
	MP	1330	1361	608	2021	5031
	Netplankton	202	285	270	116	1031
	Total activity	60635	61707	42641	39941	37711
	Dissolved (%)	33	44	71	57	64
	Particulate (%)	67	56	29	43	36
	% of time 0	100	102	70	66	62

Table A.2.12. ^{65}Zn in the water column (dpm/L); 1981 limnocorral experiment.

Limnocorral	Fraction	Date (day of year)				
		July 3 (184)	July 7 (188)	July 19 (200)	Aug 2 (214)	Aug 23 (235)
C81-1	Charcoal	2964	3496	0	3035	1299
	Anion	182	159	164	138	114
	Cation	591	866	114	121	130
	Sulfide	47	0	0	182	64
	GFC	8543	6337	1830	0	0
	MP	299	531	382	0	0
	Netplankton	42	82	23	0	25
	Total activity	12669	11471	2513	3476	1631
	Dissolved (%)	30	39	11	100	98
	Particulate (%)	70	61	89	0	2
	% of time 0	100	91	20	27	13
LM81-2	Charcoal	7119	4229	1634	733	323
	Anion	106	191	122	116	135
	Cation	2832	855	332	188	134
	Sulfide	42	0	0	48	34
	GFC	3419	2102	781	0	0
	MP	105	76	1820	162	55
	Netplankton	16	25	20	0	48
	Total activity	13639	7476	4709	1247	729
	Dissolved (%)	74	71	44	87	86
	Particulate (%)	26	29	56	13	14
	% of time 0	100	55	35	9	5
PN81-3	Charcoal	2044	2139	1191	816	564
	Anion	168	588	279	273	275
	Cation	350	288	91	134	152
	Sulfide	55	0	0	328	33
	GFC	7596	6623	3850	0	0
	MP	480	388	261	171	0
	Netplankton	73	179	73	0	102
	Total activity	10767	10203	5744	1722	1126
	Dissolved (%)	24	30	27	90	91
	Particulate (%)	76	70	73	10	9
	% of time 0	100	95	53	16	10
LS81-4	Charcoal	1481	1538	663	418	275
	Anion	341	277	125	125	121
	Cation	916	368	59	89	55
	Sulfide	29	21	0	0	54
	GFC	7338	6014	1234	0	0
	MP	879	589	407	530	0
	Netplankton	30	84	20	0	38

Table A.2.12. Cont'd.

Limnocorral	Fraction	Date (day of year)				
		July 3 (184)	July 7 (188)	July 19 (200)	Aug 2 (214)	Aug 23 (235)
	Total activity	11013	8891	2507	1163	543
	Dissolved (%)	25	25	34	54	93
	Particulate (%)	75	75	66	46	7
	% of time 0	100	81	23	11	5
C81-5	Charcoal	1845	2175	1391	1221	813
	Anion	338	254	202	73	200
	Cation	989	257	132	100	166
	Sulfide	63	0	0	45	70
	GFC	5754	5657	2720	0	0
	MP	371	496	214	373	0
	Netplankton	26	80	55	0	36
	Total activity	9385	8919	4714	1813	1284
	Dissolved (%)	34	30	37	79	97
	Particulate (%)	66	70	63	21	3
	% of time 0	100	95	50	19	14
HS81-6	Charcoal	1654	1098	521	307	200
	Anion	220	180	227	88	82
	Cation	570	427	189	88	84
	Sulfide	0	0	0	70	88
	GFC	7952	6098	1891	0	0
	MP	1909	1804	977	818	0
	Netplankton	28	90	17	0	26
	Total activity	12332	9698	3823	1370	480
	Dissolved (%)	20	18	25	40	94
	Particulate (%)	80	82	75	60	6
	% of time 0	100	79	31	11	4
C81-7	Charcoal	3220	2073	1498	1252	730
	Anion	182	491	132	132	248
	Cation	371	366	82	88	145
	Sulfide	0	0	0	57	59
	GFC	5971	6813	2413	0	0
	MP	338	327	182	314	284
	Netplankton	0	139	100	0	330
	Total activity	10082	10209	4407	1843	1796
	Dissolved (%)	37	29	39	83	66
	Particulate (%)	63	71	61	17	34
	% of time 0	100	101	44	18	18

Table A.2.13. ^{22}Na in the water column (dpm/L); 1981 limnocorral experiment.

Limnocorral	Fraction	Date (day of year)				
		July 3 (184)	July 7 (188)	July 19 (200)	Aug 2 (214)	Aug 23 (235)
LS81-4	Charcoal	0	0	0	52	0
	Anion	0	0	0	0	0
	Cation	1261	1359	1214	0	1277
	Sulfide	0	0	0	0	0
	GFC	0	0	0	0	0
	MP	0	0	0	0	0
	Netplankton	0	0	0	0	0
	Total activity	1261	1359	1214	52	1277
	Dissolved (%)	100	100	100	100	100
	Particulate (%)	0	0	0	0	0
	% of time 0	100	108	96	4	101

Appendix 3

^{203}Hg and total mercury concentrations in fish from limnocorrals, 1981-1984

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Table A.3.1. "Time zero" yellow perch and emerald shiner data, 1981. Total mercury concentration was not determined for emerald shiner.

		Whole fish		Muscle
Date sampled	Fish no.	Total length (mm)	Round wt (g)	Total Hg (μg/g)
Yellow Perch				
June 30	1	49	1.0	
	2	55	1.3	
	3	51	1.0	
	4	54	1.2	
	5	49	0.9	
	6	51	0.9	
	7	46	0.6	
	8	50	1.0	
	9	50	1.0	
	10	50	1.1	
	11	48	0.7	
	12	51	1.0	
	13	53	1.0	
	14	51	1.0	
	15	49	0.7	
	16	51	1.0	
	17	50	1.0	
	18	52	1.1	
	19	no data		
	20	55	1.2	
	21	50	1.0	
	22	53	1.1	
	23	53	1.2	
	24	52	1.1	
	25	54	1.1	
	26	53	1.2	
	27	55	1.3	
	28	56	1.4	
	29	51	1.1	
	30	51	1.1	
	31	53	1.4	
	32	55	1.3	
	33	47	0.8	
	34	49	1.0	
	35	50	1.1	
	36	44	0.6	
	37	54	1.2	
	38	53	1.3	
July 1	39	52	1.0	0.090
	40	49	0.9	
	41	52	0.9	
	42	49	0.6	
	43	50	1.0	
	44	51	1.1	
	45	54	1.2	
Yellow Perch				
July 1	46	46	0.7	
	47	48	0.8	
	48	48	0.8	
	49	49	0.9	
	50	49	0.9	
	51	53	1.3	0.090
	52	55	1.3	
	53	50	0.9	
	54	51	1.1	
	55	54	1.2	
	56	49	0.8	
	57	49	0.8	
	58	51	0.9	
	59	47	0.6	
	60	51	0.9	
	61	48	0.8	
	62	51	1.1	
	63	49	0.7	
	64	53	1.1	
	65	54	1.3	0.130
	66	51	0.9	
	67	51	1.1	
	68	49	0.7	
	69	48	0.7	
	70	53	1.2	
	71	50	1.0	0.080
	72	50	1.1	0.070
	73	47	0.7	
	74	53	1.2	
	75	52	1.1	0.090
	76	48	1.0	
	77	49	1.0	
	78	54	1.5	
	79	47	0.6	
	80	52	1.1	
	81	53	1.4	
	82	53	1.2	
	83	54	1.2	
	84	47	1.0	
	85	49	0.9	
	86	51	1.1	
	87	54	1.4	
	88	50	1.0	
	89	50	1.0	
	90	47	0.8	
Yellow Perch				
July 1	91	54	1.2	
	92	49	1.0	
	93	51	0.9	
	94	49	1.0	
	95	52	1.1	
	96	49	1.0	
	97	47	0.9	
	98	49	0.9	
	99	55	1.3	
	100	50	0.9	
July 4	101	54	1.4	
	102	49	1.0	
	103	55	1.4	
	104	49	0.7	
	105	53	1.2	
	106	52	1.2	
	107	47	0.8	
	108	56	1.5	
	109	52	1.2	
	110	50	1.0	
	111	54	1.3	
	112	54	1.1	
	113	52	1.1	
	114	53	1.1	
Emerald Shiner				
June 30	1	75	3.0	
	2	76	2.6	
	3	82	3.2	
	4	77	3.2	
	5	81	3.7	
	6	76	2.7	
	7	72	2.4	
	8	71	2.3	
	9	74	2.5	
	10	74	2.2	
	11	80	2.7	
	12	76	2.4	
	13	69	2.1	
	14	78	2.5	
	15	79	2.6	
	16	76	2.6	
	17	68	1.9	
Emerald Shiner				
June 30	18	78	2.7	
	19	74	2.2	
	20	76	2.7	
	21	75	2.5	
	22	74	2.3	
	23	79	3.0	
	24	75	2.6	
	25	84	3.7	
	26	75	2.2	
	27	76	2.5	
	28	78	2.9	
	29	69	2.0	
	30	72	2.1	
	31	77	2.2	
	32	76	2.5	
	33	68	2.0	
	34	72	1.8	
	35	77	2.4	
	36	79	2.7	
	37	79	2.9	
	38	75	2.2	
	39	74	2.2	
	40	74	2.1	
	41	72	2.2	
	42	78	2.6	
	43	77	2.7	
	44	75	2.1	
	45	78	2.9	
	46	72	2.2	
	47	75	2.2	
	48	74	2.4	
	49	67	1.8	
	50	81	3.0	
July 4	51	75	2.7	
	52	78	3.7	
	53	78	2.9	
	54	79	3.5	
	55	83	3.3	
	56	76	2.6	
	57	74	2.4	
	58	78	3.1	
	59	69	2.2	
	60	83	4.2	
	61	75	2.7	

Table A.3.2. Mercury content of yellow perch and emerald shiners from C81-1, a sediment-bottomed control limnocorral, 1981.

Species	Date(s) sampled	Fish no.	Whole fish		Whole fish less gut				Muscle tissue							Carcass			Combined muscle/carcass	
			Ttl lgth (mm)	Rnd Wt (g)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	Total Hg (μg/g)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	Total Hg (μg/g)	Specific activity (dpm/μg)	Methyl Hg (dpm/g)	Methyl Hg(% of total)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)
Yellow Perch	July 10,11	001	55	1.6	1.51	821	20987	0.043	These fish not filleted											
		002	54	1.4	1.29	500	12781	0.053												
		003	57	1.7	1.51	510	13037	0.040												
		004	52	1.2	1.04	581	14852	0.044												
		005	50	1.0	1.88	812	20757	0.023												
		006	54	1.5	1.41	667	17050	0.055												
	July 24,26	043	67	3.0	2.83	476	12168	0.033												
		044	62	2.5	2.33	552	14110	0.032												
		045	61	2.5	2.23	525	13420	0.029												
		046	65	2.8	2.63	437	11171	0.039												
		047	61	2.4	2.18	535	13676	0.026												
	Aug 2,3	085		3.9					1.82	208	5317	0.027	196926			1.68	653	16692	422	10787
		087	75	4.3					2.01	330	8436			7977	95	1.83	704	17996	508	12986
		089		4.2					2.02	337	8615	0.036	239306			1.76	648	16590	482	12321
		091		3.1					1.30	340	8691	0.019	457421			1.35	439	11222	390	9969
		093	75	4.5					1.98	353	9024	0.016	564000			2.07	715	18277	538	13753
		095	79	3.6					1.72	498	12730			10310	81	1.78	866	22137	685	17510
		097		3.8					1.72	314	8027	0.078	102910			1.69				
		099		3.9					1.87	370	9458			9076	96	1.59	499	12756	429	10966
		101		3.2					1.44	384	9816	0.029	338483			1.44	530	13548	457	11682
	Aug 24,25	300	75	4.5					1.93	482	12321			8966	73	2.26	462	11810	471	12040
		302	87	6.9					2.77	460	11759			12257	104	3.64	417	10660	436	11145
		304	85	7.7					3.13	477	12193			12134	100	3.97	446	11401	460	11759
Emerald Shiner	Aug 24,25	378	82	4.0					1.96	455	11631					1.39	689	17612	552	14110
		380	81	4.1					1.81	359	9177					1.41	594	15184	462	11810
		382	80	4.2					1.89	598	15286			18700	122	1.25	691	17664	635	16232
		384	81	4.1					1.87	479	12244					1.39	857	21907	640	16360
		386	76	3.7					1.60	414	10583					1.33	649	16590	521	13318
		388	82	4.3					1.87	413	10583					1.44	729	18635	551	14085
		390	78	3.5					1.65	611	15619					1.41	891	22776	740	18916
		392	80	4.4					1.83	486	12423					1.83	424	10838	455	11631
		394	78	4.1					1.67	477	12193			13339	109	1.52	836	21370	648	16564
		396	81	4.1					1.14	785	20066					1.54	544	13906	647	16539
		398	76	3.8					1.85	471	12040			14965	124	1.26	685	17510	558	14264
		400							1.90	751	19197					1.58	751	19197	751	19197
		402	73	3.3					1.55	461	11784					1.13	915	23390	652	16667
		404	80	4.6					1.14	798	20399					1.59	488	12474	617	15772

Note: Total length not taken when caudal fin was broken.

Table A.3.3. Mean mercury content of yellow perch and emerald shiners from C81-1, a sediment-bottomed control limnocorral, 1981. (For sample sizes, see Table A.3.2).

			Whole fish		Whole fish less gut				Muscle tissue							Carcass			Combined muscle/carcass	
Species	Date(s) sampled	Fish no.	Ttl lgth (mm)	Rnd Wt (g)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	Total Hg (μg/g)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	Total Hg (μg/g)	Specific activity (dpm/μg)	Methyl Hg (dpm/g)	Methyl Hg(% of total)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)
Yellow Perch	June 30-July 4	Time 0	51	1.0								0.090								
	July 10,11	001-006	54	1.4	1.4	649	16577	0.043												
	July 24,26	043-047	63	2.6	2.4	505	12909	0.032												
	Aug 2,3	085-101	76	3.8					1.76	348	8902	0.034	316508	9121	91	1.69	632	16152	489	12497
	Aug 24,25	300-304	82	6.4					2.61	473	12091			11119	92	3.29	442	11290	456	11648
Emerald Shiner	June 30-July 4	Time 0	76	2.6																
	Aug 24,25	378-404	79	4.0					1.70	540	13802			15668	118	1.43	696	17790	602	15390

Table A.3.4. Mercury content of yellow perch and emerald shiners from LM81-2, a sediment-bottomed limnocorral with added (low) moss-peat, 1981.

Species	Date(s) sampled	Fish no.	Whole fish		Whole fish less gut				Muscle tissue							Carcass			Combined muscle/carcass	
			Ttl lgth (mm)	Rnd Wt (g)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	Total Hg (μg/g)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	Total Hg (μg/g)	Specific activity (dpm/ug)	Methyl Hg (dpm/g)	Methyl Hg(% of total)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)
Yellow Perch	July 10,11	007	56	1.7	1.5	567	14494	0.054												
		008	53	1.4	1.3	876	22393	0.034												
		009	54	1.5	1.4	413	10557	0.032												
		010	55	1.3	1.2	656	16769	0.049												
		011	55	1.2	1.1	306	7822	0.050												
		012	54	1.6	1.4	729	18635	0.056	These fish not filleted											
	July 24,26	049	69	3.5	3.1	4126	105470	0.142												
		050	55	1.6	1.4	3607	92203	0.144												
		051	70	3.6	3.1	2930	74898	0.129												
		052	58	1.7	1.8	3840	98160	0.101												
		053	61	2.1	2.0	3022	77249	0.124												
		054	67	3.1	2.8	3856	98569	0.123												
	Aug 2,3	103		2.9					1.19	11400	291411	0.220	1324595			1.21	9737	248901	10562	269990
		105	68	3.3					1.38	11773	300946					1.50	10735	274412	11232	287117
		107	77	5.0					2.13	7185	183666					2.20	6020	153885	6593	168533
		109		3.3					1.45	9956	254499					1.47	8792	224744	9370	239519
		111	68	2.8					1.25	8304	212270					1.31	7492	191513	7888	201636
		113		3.4					1.47	12260	313395			304005	97	1.54	10210	260992	11211	286580
		115	68	3.6					1.51	10931	279422					1.60	8550	218558	9706	248108
		117		3.7					1.65	8533	218124			223769	103	1.68	7866	201074	8196	209509
		119		4.6					2.02	8080	206544					2.01	7014	179294	7548	192945
		121	69	3.0					1.26	6720	171779			169123	98	1.29	5792	148057	6251	159790
		123	70	3.8					1.67	8196	209509					1.66	8906	227658	8550	218558
		125		3.1					1.30	9391	240056					1.43	8713	222725	9036	230982
		127	68	3.3					1.41	7598	194223	0.170	1142488			1.49	6840	174847	7209	184279
		129		3.7					1.53	7539	192715	0.170	1133618			1.79	7693	196651	7622	194836
		131		3.3	2.9	7888	201636													
		133	69	3.5					1.47	9201	235199	0.200	1175995			1.70	8902	227556	9041	231109
		135	70	4.2					1.47	9211	235455	0.210	1121214			1.58	8776	224335	8986	229703
		137	67	2.9					1.34	11794	301483	0.250	1205932			1.25	10877	278042	11351	290158
	Aug 24,25	306	83	5.6					2.26	27304	697955	0.610	1144189			2.78	18933	483972	22687	579934
		308	93	9.1					4.11	16249	415363	0.360	1153786			4.46	11693	298901	13878	354755
		310	80	5.7					2.52	16907	432183	0.400	1080458			2.66	12002	306800	14388	367791
		312	78	5.2					1.97	28155	719709	0.460	1564585			2.76	18570	474693	22562	576738
		314	78	5.4					2.13	21740	555726	0.400	1389315			2.71	16767	428604	18956	484560
		316	79	5.1					2.23	25365	648390	0.500	1296780			2.29	18824	481186	22051	563676
		318	85	6.2					2.84	22140	565951					2.66	16546	422955	19435	496805
		320	75	5.0					2.27	27045	691334					2.06	19998	511196	23692	605624
		322	79	5.0					2.05	26154	668558					2.09	17745	453604	21909	560046
		324	79	5.6					2.51	22625	578349					2.29	17269	441437	20070	513037
		326	78	4.5					1.98	31048	793661			890255	112	2.01	21840	558282	26409	675077
		328	78	4.8					2.09	25542	652914			619791	95	1.95	19435	496805	22594	577556
		330	77	5.0					2.35	28926	739417	0.570	1297223			1.96	19196	490695	24501	626304
		332	85	6.7					2.85	19058	487168			414436	85	2.89	13555	346498	16287	416334
		334	78	4.6					1.98	27642	706595	0.550	1284718			1.93	20192	516155	23965	612602
		336	81	5.5					2.36	21271	543737	0.460	1182037			2.29	12845	328349	17121	437653
Emerald Shiner	Aug 24,25	406	78	4.1					1.81	22539	576150			565629	98	1.69	15869	405649	19318	493814
		408	78	3.8					2.05	26316	672699					1.24	17218	440133	21627	552837
		410	81	4.6					2.18	20362	520501					1.89	17236	440593	18910	483384
		412	81	4.3					2.26	27334	698722					1.45	18283	467357	23778	607822
		414	79	4.3					1.98	22616	578119					1.82	18236	466155	20518	524489
		416	76	3.6					1.79	19453	497265			506623	102	1.33	11134	284611	15907	406621
		418	85	5.7					1.65	23460	599693					2.00	11393	291232	16848	430675
		420	73	5.0					2.33	25927	662756					1.65	18330	468558	22777	582234
		424	75	3.4					1.53	32025	818635					1.33	18629	476201	25795	659381
		426	80	4.2					2.16	18128	463395					1.19	13507	345271	16487	421447
		428	77	4.1					1.85	23125	591130			634347	107	1.40	17295	442101	20614	526943

Note: Total length not taken when caudal fin was broken.

Table A.3.5. Mean mercury content of yellow perch and emerald shiners from LM81-2, a sediment-bottomed limnocorral with added (low) moss-peat, 1981. (For sample sizes, see Table A.3.4).

			Whole fish		Whole fish less gut				Muscle tissue							Carcass			Combined muscle/carcass	
Species	Date(s) sampled	Fish no.	Ttl lgth (mm)	Rnd wt (g)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	Total Hg (µg/g)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	Total Hg (µg/g)	Specific activity (dpm/µg)	Methyl Hg (dpm/g)	Methyl Hg(% of total)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)
Yellow Perch	June 30-July 4	Time 0	51	1.0								0.090								
	July 10,11	007-012	55	1.5	1.3	591	15112	0.046												
	July 24,26	049-054	63	2.6	2.4	3564	91092	0.127												
	Aug 2,3	103-137	69	3.5					1.50	9298	237688	0.203	1183974	232299	99	1.57	8407	214897	8844	226080
	Aug 24,25	306-336	80	5.6					2.41	24198	618563	0.479	1265899	641494	97	2.49	17213	440008	20657	528031
Emerald Shiner	June 30-July 4	Time 0	76	2.6																
	Aug 24,25	406-428	78	4.3					1.96	23753	607188			568866	102	1.54	16103	411624		

Table A.3.6. Mercury content of yellow perch and emerald shiners from PN81-3 , a plastic bottomed limnocorral with added phosphorus and nitrogen, 1981.

Species	Date(s) sampled	Fish no.	Whole fish		Whole fish less gut				Muscle tissue							Carcass			Combined muscle/carcass	
			Ttl Lgth (mm)	Rnd Wt (g)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	Total Hg (µg/g)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	Total Hg (µg/g)	Specific Activity (dpm/ug)	Methyl Hg (dpm/g)	Methyl Hg(% of total)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)
	July 10,11	013	53	1.3	1.2	996	25460	0.071	These fish not filleted											
		014	56	1.5	1.5	838	21421	0.069												
		015	56	1.5	1.4	709	18124	0.052												
		016	54	1.3	1.3	860	21984	0.041												
		017	56	1.5	1.4	757	19351	0.064												
		018	54	1.4	1.3	828	21166	0.041												
	July 24,26	055	58	1.6	1.6	712	18200	0.041												
		056	59	1.8	1.7	669	17101	0.035												
		057	54	1.3	1.3	661	16897	0.042												
		058	67	2.9	2.7	758	19376	0.033												
		059	56	1.4	1.3	853	21805	0.034												
		060	64	2.3	2.1	733	18737	0.054												
	Aug 2,3	141	63	2.7					1.05	705	18021			16644	92	1.12	1456	37219	1093	27940
		143	65	2.6					1.13	681	17408	0.041	424585			1.28	991	25332	846	21626
		145	68	2.9					1.28	757	19351			15344	79	1.26	1159	29627	956	24438
		147	60	2.1					0.92	687	17561	0.045	390244			0.88	1123	28707	900	23006
		149		2.0					0.93	1268	32413	0.033	982212			0.88	1181	30189	1226	31339
		151		2.5					1.24	601	15363			10102	66	1.15	870	22239	730	18661
		153		2.6					0.99	1007	25741	0.046	559587			1.21	1261	32234	1147	29320
		155		2.6					1.14	774	19785	0.040	494625			1.09	1398	35736	1079	27582
		157		2.4					1.00	696	17791	0.043	413744			0.99	1435	36682	1064	27198
		159	65	2.6					1.22	796	20348					1.06	1146	29294	959	24514
	Aug 24,25				No Sample															
Emerald Shiner	Aug 24,25	430	85	4.0					1.91	1215	31058					1.58	1573	40210	1377	35199
		434	86	4.6					2.17	1253	32030			30076	94	1.78	1506	38497	1367	34944
		436	83	3.8					1.69	1269	32439			30628	94	1.44	1414	36145	1336	34151
		438	80	3.6					1.53	1841	47060			41853	89	1.55	1600	40900	1720	43967

Note: Total length not taken when caudal fin was broken.

Table A.3.7. Mean mercury content of yellow perch and emerald shiners from PN81-3 , a plastic-bottomed limnocorral with added phosphorus and nitrogen, 1981. (For sample sizes, see Table A.3.6).

			Whole fish		Whole fish less gut				Muscle tissue							Carcass			Combined muscle/carcass	
Species	Date(s) sampled	Fish no.	Ttl lgth (mm)	Rnd Wt (g)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	Total Hg (µg/g)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	Total Hg (µg/g)	Specific activity (dpm/µg)	Methyl Hg (dpm/g)	Methyl Hg(% of total)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)
Yellow Perch	June 30- July 4	Time 0	51	1.0																
	July 10,11	013-018	55	1.4	1.3	831	21251	0.056				0.090								
	July 24,26	055-060	60	1.9	1.8	731	18686	0.040												
	Aug 2,3	141-159	64	2.5																
	Aug 24,25					No Sample			1.09	797	20378	0.041	544166	14030	79	1.09	1202	30726	1000	25562
Emerald Shiner	June 30- July 4	Time 0	76	2.6																
	Aug 24,25	430-438	84	4.0					1.83	1395	35647			34186	92	1.59	1523	38938	1450	37065

Table A.3.8. Mercury content of yellow perch and emerald shiners from LS81-4, a plastic-bottomed limnocorral with added (low) bank clay, 1981.

Species	Date(s) sampled	Fish no.	Whole fish		Whole fish less gut				Muscle tissue							Carcass			Combined muscle/carcass	
			Ttl lgth (mm)	Rnd wt (g)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	Total Hg (μg/g)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	Total Hg (μg/g)	Specific activity (dpm/μg)	Methyl Hg (dpm/g)	Methyl Hg(% of total)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)
Yellow Perch	July 10,11	019	56	1.6	1.41	876	22393	0.061												
		020	58	1.7	1.59	538	13753	0.060												
		021	56	1.5	1.42	539	13778	0.057												
		022	56	1.8	1.59	752	19223	0.060												
		023	57	1.3	1.19	505	12909	0.058												
		024	52	1.4	1.21	507	12960	0.035	These fish not filleted											
	July 24,26	061	64	2.4	2.15	1671	42715	0.049												
		062	60	1.9	1.64	1184	30266	0.026												
		063	61	2.1	1.89	651	16641	0.043												
		064	64	2.6	2.17	1299	33206	0.048												
		065	62	2.2	1.89	583	14903	0.028												
		066	65	2.6	2.29	797	20373	0.035												
	Aug 2,3	205		2.6					0.92	668	17076					0.96	1084	27710	880	22495
		207	60	2.2					0.95	460	11759	0.049	239980			0.81	1043	26662	728	18609
		217		2.8					1.33	2609	66692	0.054	1235037			1.01	2597	66385	2604	66564
		219	65	3.2					1.40	2101	53707	0.042	1278738			1.23	2412	61656	2246	57413
		221		3.1					1.33	1260	32209	0.044	732023			1.41	1399	35762	1332	34049
		223	68	3.3					1.53	1980	50613	0.043	1177047			1.33	2084	53272	2028	51840
		225		2.7					1.28	493	12602					0.97	833	21293	640	16360
		229	64	2.7					1.14	691	17664	0.033	535273			1.07	981	25077	831	21242
		231	65	2.7					1.26	679	17357					1.02	1009	25792	827	21140
		233	64	2.7					1.00	334	8538					1.15	984	25153	682	17434
		235	63	2.6					1.19	1175	30036					1.08	1071	27377	1126	28783
		237		2.8					1.34	1245	31825					1.11	1486	37986	1354	34611
		239	66	3.0					1.33	709	18124			14119	78	1.14	1022	26125	853	21805
		241	64	2.5					1.13	493	12602					1.01	1012	25869	738	18865
		243	66	2.6					1.23	1057	27019			24780	92	1.02	1278	32669	1157	29576
		245	67	3.3					1.50	1353	34586					1.29	1932	49387	1621	41437
		247	60	2.7					1.11	1014	25920					1.06	1214	31033	1112	28425
		249	71	4.2					2.01	1299	33206			29920	90	1.52	1536	39264	1401	35813
	Aug 24,25	338	69	3.6					1.42	2418	61810			55450	90	1.73	2083	53246	2234	57106
		340	79	4.7					2.05	1879	48032			49333	103	2.50	1281	32745	1550	39622
		342	85	6.5					2.48	1560	39877			42760	107	3.37	1363	34842	1447	36989
		346	83	5.7					2.07	2794	71421	0.044	1623205			2.91	2856	73006	2830	72342
Emerald Shiner	Aug 24,25	440	78	3.9					1.76	3428	87628					1.13	2417	61784	3033	77531
		442	82	5.1					2.35	1958	50051			49914	98	1.61	1943	49668	1952	49898
		444	77	3.2					1.46	2562	65491					1.31	2286	58436	2431	62142
		446	81	4.1					1.89	3479	88931					1.45	2693	68839	3138	80215
		448	84	4.4					1.96	2342	59867			64375	108	1.80	2502	63957	2419	61835
		450	83	4.1					1.89	2909	74361					1.48	2383	60915	2678	68456
		452	75	3.3					1.42	3459	88420			91057	103	1.16	2686	68661	3111	79525
		454	78	3.9					1.51	3435	87807					1.75	2552	65235	2961	75690

Note: Total length not taken when caudal fin was broken.

Table A.3.9. Mean mercury content of yellow perch and emerald shiners from LS81-4, a plastic-bottomed limnocorral with added (low) bank clay, 1981. (For sample sizes, see Table A.3.8).

Species	Date(s) sampled	Fish no.	Whole fish		Whole fish less gut				Muscle tissue							Carcass			Combined muscle/carcass	
			Ttl lgth (mm)	Rnd Wt (g)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	Total Hg (μg/g)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	Total Hg (μg/g)	Specific activity (dpm/μg)	Methyl Hg (dpm/g)	Methyl Hg(% of total)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)
Yellow Perch	June 30-July 4	Time 0	51	1.0								0.090								
	July 10,11	019-024	56	1.6	1.40	620	15836	0.055												
	July 24,26	061-066	63	2.3	2.01	1031	26351	0.038												
	Aug 2,3	205-249	65	2.9					1.28	1090	27863	0.044	866350	22940	87	1.12	1388	35471	1231	31470
	Aug 24,25	338-346	79	5.1					2.01	2163	55285	0.044	1623205	49181	100	2.63	1896	48460	2015	51515
Emerald Shiner	June 30-July 4	Time 0	76	2.6																
	Aug 24,25	440-454	80	4.0					1.78	2947	75320			68449	103	1.46	2433	62187	2715	69412

Table A.3.10. Mercury content of yellow perch and emerald shiners from C81-5, a plastic-bottomed control limnocorral, 1981.

Species	Date(s) sampled	Fish no.	Whole fish		Whole fish less gut				Muscle tissue							Carcass			Combined muscle/carcass	
			Ttl lgth (mm)	Rnd Wt (g)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	Total Hg (μg/g)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	Total Hg (μg/g)	Specific activity (dpm/μg)	Methyl Hg (dpm/g)	Methyl Hg(% of total)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)
Yellow Perch	July 10,11	025	55	1.4	1.29	621	15874	0.038												
		026	56	1.6	1.45	794	20297	0.071												
		027	52	1.2	1.07	932	23824	0.056												
		028	56	1.4	1.25	696	17791	0.043												
		029	56	1.5	1.37	861	22009	0.030												
		030	50	0.9	0.78	1460	37321	0.055	These fish not filleted											
	July 24,26	067	61	2.2	1.89	694	17740	0.045												
		068	61	2.1	1.79	634	16207	0.020												
		069	60	2.2	1.93	788	20143	0.025												
		070	58	1.9	1.62	932	23824	0.026												
		071	59	2.0	1.72	858	21933	0.022												
		072	60	2.1	1.83	915	23390	0.055												
	Aug 2,3	161	59	2.2					0.88	875	22367	0.056	399411			0.86	1362	34816	1116	28528
		163	68	3.5					1.58	713	18226	0.026	701000			1.33	1148	29346	912	23313
		165	66	3.2					1.40	787	20118	0.035	574800			1.12	1061	27122	909	23236
		167	73	4.3					1.87	832	21268			15330	72	1.56	1157	29576	980	25051
		169	69	3.3					1.43	745	19044	0.029	656690			1.31	1119	28604	924	23620
		171	70	3.5					1.49	989	25281	0.052	486173			1.33	1589	40619	1272	32515
		173	69	3.7					1.61	796	20348			11519	57	1.42	1249	31927	1008	25767
		175	67	3.2					1.43	1070	27352					1.21	1765	45118	1389	35506
		177	67	3.1					1.23	846	21626					1.11	1143	29218	987	25230
		179	73	4.2					1.83	1489	38062					1.62	1570	40133	1527	39034
		181	67	3.1					1.34	851	21754			12019	55	1.25	1568	40082	1197	30598
		183	66	3.0					1.23	651	16641					1.17	1209	30905	923	23594
		185	68	3.4					1.48	1098	28067	0.038	738605			1.19	1514	38701	1283	32797
		187	67	3.2					1.32	782	19990					1.14	1025	26201	895	22878
		189	69	3.7					1.69	996	25460					1.35	1140	29141	1060	27096
		191		3.1					1.32	817	20884					1.06	1342	34305	1051	26866
		193	66	3.1					1.46							1.13	1084	27710		
		195	63	2.9					1.05	891	22776					1.05	1452	37117	1172	29959
	Aug 24,25	348	82	6.2					2.98	1011	25844	0.110	234945			3.12	1366	34918	1193	30496
		350	78	4.7					2.02	1146	29294	0.100	292940			2.45	1600	40900	1395	35660
		352	83	4.3					2.07	828	21166	0.070	302371			3.00	1310	33487	1113	28451
		354	84	6.6					2.24	1045	26713			22725	85	4.20	1854	47393	1573	40210
		356	81	5.6					2.35	726	18558			14905	80	2.70	1291	33001	1028	26278
		358	86	6.9					2.37	707	18073	0.032	564781			3.84	1012	25869	896	22904
		360	86						2.39	1422	36350	0.044	826136			3.91	1476	37730	1456	37219
		362	78	5.8					2.27	1159	29627			21342	72	3.09	1651	42203	1443	36887
		364	77	4.7					1.95	971	24821	0.045	551578			2.32	1334	34100	1168	29857
		366	85	6.1					2.43	678	17331	0.035	495171			2.94	1151	29422	937	23952
Emerald Shiner	Aug 24,25	456	75	3.2					1.53	1314	33589					1.32	1415	36171	1361	34790
		458	78	3.9					1.72	1295	33103					1.61	1987	50792	1630	41667
		460	76	3.3					1.72	1358	34714					1.15	2345	59944	1753	44811
		462	78	3.9					1.79	1358	34714			26920	78	1.52	2044	52249	1673	42766
		464	77	3.5					1.66	1393	35608			28807	81	1.06	2376	60736	1776	45399
		466	82	4.7					2.34	1302	33282					1.32	1807	46191	1484	37935
		468	81	3.8					1.62	1398	35736			28287	79	1.53	1752	44785	1570	40133
		470	84	4.6					2.12	1650	42178					1.44	1765	45118	1697	43379
		472	83	4.3					2.20	1241	31723					1.27	1587	40567	1368	34969
		474	80	4.0					2.01	1315	33615					1.37	1758	44939	1495	38216
		476	77	3.6					1.68	1409	36017					1.32	1310	33487	1365	34893

Note: Total length not taken when caudal fin was broken.

Table A.3.11. Mean mercury content of yellow perch and emerald shiners from C81-5, a plastic-bottomed control limnocorral, 1981. (For sample sizes, see Table A.3.10).

			Whole fish		Whole fish less gut				Muscle tissue							Carcass			Combined muscle/carcass	
Species	Date(s) sampled	Fish no.	Ttl lgth (mm)	Rnd wt (g)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	Total Hg (μg/g)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	Total Hg (μg/g)	Specific activity (dpm/μg)	Methyl Hg (dpm/g)	Methyl Hg(% of total)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)
Yellow Perch	June 30- July 4	Time 0	51	1.0																
	July 10,11	025-030	54	1.3	1.20	894	22853	0.049				0.090								
	July 24,26	067-072	60	2.1	1.80	804	20540	0.032												
	Aug 2,3	161-195	67	3.3					1.42	896	22898	0.039	592780	12956	61	1.23	1305	33369	1094	27976
	Aug 24,25	348-366	82	5.1					2.31	969	24778	0.062	466846	19657	79	3.16	1405	35902	1220	31191
Emerald Shiner	June 30- July 4	Time 0	76	2.6																
	Aug 24,25	456-476	79	3.9					1.85	1367	34934			28005	79	1.36	1831	46816	1561	39905

Table A.3.12. Mercury content of yellow perch and emerald shiners from HS81-6, a plastic-bottomed limnocorral with added (high) bank clay, 1981.

Species	Date(s) sampled	Fish no.	Whole fish		Whole fish less gut				Muscle tissue							Carcass			Combined muscle/carcass	
			Ttl lgth (mm)	Rnd Wt (g)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	Total Hg (µg/g)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	Total Hg (µg/g)	Specific activity (dpm/µg)	Methyl Hg (dpm/g)	Methyl Hg(% of total)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)
Yellow Perch	July 10,11	031	55	1.5	1.33	839	21447	0.043	These fish not filleted											
		032	48	1.0	0.91	596	15235	0.061												
		033	51	1.2	1.13	614	15695	0.035												
		034	53	1.2	1.15	671	17152	0.055												
		035	58	1.7	1.56	585	14954	0.044												
		036	52	1.4	1.18	759	19402	0.051												
	July 24,26	073	65	2.3	1.96	614	15695	0.039												
		073	60	1.9	1.67	751	19197	0.039												
		075	60	2.0	1.76	652	16667	0.019												
		076	62	2.1	1.83	563	14392	0.037												
		077	60	2.1	1.82	755	19300	0.038												
		078	60	2.0	1.73	608	15542	0.056												
	Aug 2,3	197		3.0					1.22	610	15593			17066	109	1.40	652	16667	632	16155
		199	69	3.3					1.49	1115	28502					1.23	1275	32592	1187	30343
		201	61	2.7					1.00	564	14417					0.97	821	20987	691	17664
		203	62	2.5					0.98	608	15542	0.038	409000			0.93	1068	27301	832	21268
		272		2.9					1.32	572	14622			15496	106	1.40	1187	30343	889	22725
		274		2.2					0.97	634	16207					0.95	2028	51840	1324	33845
		276	68	3.0					1.25	1120	28630					1.28	602	15389	858	21933
		278	72	3.8					1.58	489	12500					1.78	766	19581	636	16258
		280		2.6					1.10	533	13625	0.042	324405			1.16	883	22572	713	18226
		282	66						1.23	498	12730	0.044	289318			1.32	718	18354	612	15644
		284		3.0					1.33	794	20297	0.041	495049			1.26	613	15670	706	18047
		286	70	3.1					1.09	644	16462	0.031	531032			1.50	815	20833	743	18993
		288		3.0					1.47	584	14928	0.045	331733			1.25	1103	28195	823	21038
		290		4.3					1.98	583	14903			10386	70	1.98	1038	26534	811	20731
		292		3.0					1.33	581	14852					1.33	1023	26150	802	20501
		294		2.5					1.94	347	8870					1.14	760	19427	500	12781
		296		3.8					1.50	625	15976					1.63	988	25256	814	20808
		298	73	3.7					1.64	542	13855					1.64	1144	29243	843	21549
	Aug 24,25	374	88	7.8					3.30	1323	33819			34150	101	3.32	1109	28349	1216	31084
		376	76	4.8					2.23	1101	28144			28217	100	2.13	1142	29192	1121	28655
Emerald Shiner	Aug 24,25	478	77	3.2					1.67	1399	35762			34157	96	0.93	1028	26278	1266	32362
		480	72	3.1					1.60	1348	34458					1.07	1437	36733	1384	35378
		482	82	4.2					1.88	995	25435					1.48	1203	30752	1087	27786
		484	76	3.5					1.75	1316	33640					1.18	1973	50435	1581	40414
		486	75	4.0					2.01	1475	37704					1.28	1622	41462	1532	39162
		488	74	3.7					1.86	1666	42587			35026	93	1.31	1606	41053	1641	41948
		490	82	4.8					2.10	1235	31570					1.74	1596	40798	1399	35762
		492	80	3.8					1.79	1792	45808					1.29	2124	54294	1931	49361
		494	80	4.5					2.12	1458	37270			34876	94	1.60	1550	39622	1498	38292
		498	77	3.5					1.74	1883	48134					1.07	1588	40593	1771	45271
		502	77	3.3					1.43	1407	35966					1.27	2131	54473	1748	44683
		506	81	3.8					1.77	1506	38497					1.30	1213	31007	1382	35327
		508	82	4.6					1.22	1806	46166					1.49	1297	33154	1526	39008

Note: Total length not taken when caudal fin was broken.

Table A.3.13. Mean mercury content of yellow perch and emerald shiners from HS81-6, a plastic-bottomed limnocorral with added (high) bank clay, 1981. (For sample sizes, see Table A.3.12).

			Whole fish		Whole fish less gut				Muscle tissue							Carcass			Combined muscle/carcass	
Species	Date(s) sampled	Fish no.	Ttl lgth (mm)	Rnd wt (g)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	Total Hg (μg/g)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	Total Hg (μg/g)	Specific activity (dpm/μg)	Methyl Hg (dpm/g)	Methyl Hg(% of total)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)
Yellow Perch	June 30-July 4	Time 0	51	1.0								0.090								
	July 10,11	031-036	53	1.3	1.21	677	17314	0.048												
	July 24,26	073-078	61	2.1	1.80	657	16799	0.038												
	Aug 2,3	197-298	68	3.1					1.36	636	16251	0.040	396756	14316	95	1.34	971	24830	801	20473
	Aug 24,25	374-376	82	6.3					2.77	1212	30982			31184	101	2.73	1126	28771	1169	29870
Emerald Shiner	June 30-July 4	Time 0	76	2.6																
	Aug 24,25	478-508	78	3.9					1.76	1484	37923			34686	94	1.31	1567	40050	1519	38827

Table A.3.14. Mercury content of yellow perch and emerald shiners from C81-7, a plastic-bottomed control limnocorral, 1981.

			Whole fish		Whole fish less gut				Muscle tissue							Carcass			Combined muscle/carcass	
Species	Date(s) sampled	Fish no.	Ttl lgth (mm)	Rnd wt (g)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	Total Hg (µg/g)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	Total Hg (µg/g)	Specific activity (dpm/µg)	Methyl Hg (dpm/g)	Methyl Hg(% of total)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)
Yellow Perch	July 10,11	037	53	1.2	1.18	887	22674	0.033												
		038	53	1.4	1.29	924	23620	0.042												
		039	57	1.6	1.54	739	18891	0.029												
		040	52	1.2	1.09	808	20654	0.040												
		041	49	1.0	0.93	1604	41002	0.042												
		042	55	1.5	1.35	1044	26687	0.059	These fish not filleted											
	July 24,26	079	62	2.4	2.08	704	17996	0.033												
		080	58	1.9	1.69	753	19248	0.038												
		081	63	2.5	2.12	706	18047	0.044												
		082	60	2.1	1.85	633	16181	0.032												
		083	60	2.4	2.04	930	23773	0.043												
		084	58	1.8	1.54	714	18252	0.029												
	Aug 2,3	209	67	3.3					1.52	591	15107					1.28	1290	32975	911	23287
		211	61	2.6					1.02	826	21115			12648	60	0.88	1804	46115	1279	32694
		227	63	2.6					1.27	610	15593					0.99	1129	28860	837	21396
		251	61	2.6					1.02	297	7592	0.046	165043			0.97	669	17101	478	12219
		253	62	2.8					1.27	651	16641	0.044	378205			1.07	1299	33206	947	24208
		255	58	2.0					0.92	765	19555			6278	32	0.82	1620	41411	1168	29857
		257	61	2.5					1.16	640	16360	0.043	380465			0.90	1140	29141	858	21933
		259	63	2.6					1.13	466	11912	0.050	238240			0.94	1156	29550	779	19913
		261	67	3.2					1.50	523	13369	0.043	310907			1.20	975	24923	724	18507
		263	64	2.9					1.28	448	11452	0.089	128674			1.03	1022	26125	704	17996
		265	69	3.1					1.53	457	11682			4590	39	1.19	1019	26048	703	17970
		267	67	3.1					1.49	608	15542					1.19	1132	28937	841	21498
		269	66	3.2					1.54	494	12628					1.19	1095	27991	756	19325
	Aug 24,25				No sample															
Emerald Shiner	Aug 24,26	510	74	4.3					2.19	384	9816					1.37	1025	26201	631	16130
		512	75	4.0					1.71	573	14647					1.71	1460	37321	1017	25997
		514	75	3.3					1.52	567	14494					1.38	1070	27352	806	20603
		516	79	4.1					1.87	623	15925			6386	40	1.73	1400	35787	996	25460
		518	75	3.2					1.41	491	12551					1.38	1161	29678	822	21012
		520	84	4.6					2.03	669	17101					1.90	1480	37832	1061	27122
		522	72	3.6					1.37	769	19657					1.75	1484	37935	1170	29908
		524	77	3.4					1.60	685	17510					1.49	2066	52812	1351	34535
		526	75	3.4					1.47	655	16743					1.16	1390	35532	979	25026
		528	75	3.6					1.56	677	17306					1.29	2506	64059	1505	38471
		530	77	5.0					2.27	569	14545			9200	63	1.68	1454	37168	945	24156
		532	80	4.4					2.02	642	16411					1.58	1619	41385	1071	27377
		534	80	4.2					1.80	643	16437					1.58	1801	46038	1184	30266
		536	75	4.0					1.97	396	10123			4735	47	1.32	896	22904	597	15261

Note: Total length not taken when caudal fin was broken.

Table A.3.15. Mean mercury content of yellow perch and emerald shiners from C81-7, a plastic-bottomed control limnocorral, 1981. (For sample sizes, see Table A.3.14).

			Whole fish		Whole fish less gut				Muscle tissue							Carcass			Combined muscle/carcass	
Species	Date(s) sampled	Fish no.	Ttl lgth (mm)	Rnd Wt (g)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	Total Hg (μg/g)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	Total Hg (μg/g)	Specific activity (dpm/μg)	Methyl Hg (dpm/g)	Methyl Hg(% of total)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)
Yellow Perch	June 30-July 4	Time 0	51	1.0								0.090								
	July 10,11	037-042	53	1.3	1.20	1001	25588	0.041												
	July 24,26	079-084	60	2.2	1.90	740	18916	0.037												
	Aug 2,3	209-269	64	2.8																
	Aug 24,25					No Sample			1.28	567	14504	0.053	266922	7839	44	1.05	1181	30183	845	21600
Emerald Shiner	June 30-July 4	Time 0	76	2.6																
	Aug 24,25	510-536	77	3.9					1.77	596	15233			6774	50	1.52	1487	38000	1010	25809

Table A.3.16. "Time zero" data on yellow perch, 1982.

Date sampled	Fish no.	Whole fish Total length (mm)	Whole fish Round wt (g)	Muscle tissue Weight (g)	Muscle tissue Total Hg (µg/g)	Carcass Weight (g)
July 5	1	61	1.75			
	2	57	1.33			
	3	59	1.72			
	4	56	1.51			
	5	58	1.49			
	6	60	1.77			
	7	56	1.36			
	8	64	1.99			
	9	49	0.91			
	10	57	1.48			
	11	57	1.57			
	12	57	1.47			
	13	60	1.89	0.35	0.034	1.43
	14	55	1.17			
	15	53	1.07			
	16	56	1.33			
	17	62	2.05			
	18	56	1.25			
	19	50	1.00			
	20	50	0.96			
	21	57	1.46			
July 3	22	54	1.19			
	23	59	1.46			
	24	59	1.64			
	25	63	1.94	0.83	0.034	1.01
	26	58	1.67			
	27	55	1.24	0.40	0.047	0.75
	28	61	2.01	0.71	0.047	1.18
	29	55	1.23			
	30	59	1.61			
July 3	31	55	1.37			
	32	50	1.04			
	33	54	1.40			
	34	58	1.68			
	35	54	1.38	0.49	0.34	0.86
	36	56	1.67			
	37	62	2.05			
	38	57	1.53			
	39	55	1.28			
	40	55	1.19			
	41	64	2.21			
	42	62	2.11			
	43	60	1.57			
	44	53	1.08			
	45	61	1.83			
	46	54	1.14			
	47	56	1.34			
	48	49	0.86			
	49	53	1.36			
	50	56	1.38			
	51	59	1.68			
	52	51	1.01			
	53	50	1.00			
	54	49	0.80			
	55	52	0.85			
	56	55	1.17			
	57	56	1.44			
	58	58	1.32	0.39	0.059	0.89

Table A.3.17. Mercury content of yellow perch from FM82-1, a limnocorral located over a flooded moss/sedge/grass bottom, 1982.

Date(s) sampled	Fish no.	Whole fish				Muscle tissue					Carcass			Combined muscle/carcass	
		Ttl lgth (mm)	Rnd wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	Total Hg (μg/g)	Specific activity (dpm/μg)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)
July 19, 21	101	68	3.08			1.04	183	4678	0.053	88264	1.50	587	15005	422	10787
	102	64	2.65			0.92	306	7822	0.049	159633	1.22	658	16820	507	12960
	103	60	1.90			0.49	525	13420	0.066	203333	0.94	898	22955	770	19683
	104	58	1.55			0.41	384	9816	0.059	166373	0.70	802	20501	648	16564
	105	59	1.61			0.55	550	14059	0.061	230475	0.91	908	23211	773	19760
	106	63	2.23			0.59	470	12014	0.072	166861	1.10	803	20527	687	17561
Aug 2	107	65	2.49			0.96	369	9433	0.045	209622	1.15	591	15107	490	12526
	108	72	2.89			1.15	374	9560	0.061	156721	1.31	525	13420	454	11605
	109	58	1.87			0.75	856	21881	0.053	412849	0.78	808	20654	832	21268
	110	63	2.10			0.78	504	12883	0.053	243075	0.98	690	17638	608	15542
	111	60	2.30			0.82	803	20527	0.064	320734	1.12	1006	25716	920	23517
	112	64	2.80			1.05	276	7055	0.053	133113	1.28	486	12423	391	9995
Aug 16	113	72	3.48			1.23	575	14698	0.053	277321	1.61	702	17945	647	16539
	114	75	4.09			1.59	451	11529	0.045	256200	1.57	596	15235	523	13369
	115	74	3.29			1.22	943	24105	0.066	365227	1.57	903	23083	920	23517
	116	72	3.64			1.49	419	10711	0.038	281868	1.50	554	14162	487	12449
	117	70	3.38			1.22	478	12219	0.049	249367	1.58	550	14059	519	13267
	118	64	2.07			0.74	612	15644	0.053	295170	0.89	744	19018	684	17485
Aug 30	119	75	4.15			1.57	479	12244	0.027	453481	1.91	481	12296	480	12270
	120	74	4.21			1.60	661	16897	0.041	412122	1.90	577	14749	615	15721
	121	70	3.32			1.21	935	23901	0.042	569071	1.58	765	19555	839	21447
	122	75	4.23			1.64	837	21396	0.021	1018857	1.71	755	19300	795	20322
	123	78	4.83			1.77	553	14136	0.023	614609	2.05	629	16079	594	15184
	124	80	5.49			2.17	551	14085	0.013	1083462	2.36	515	13165	532	13599
	125	71	3.25	661	16897	1.11	684	17485			1.57	786	20092	744	19018
	126	66	2.63	677	17306	0.87	670	17127			1.39	655	16743	661	16897
	127	66	2.57	773	19760	0.77	995	25435			1.23	934	23875	957	24463
	128	61	2.10	811	20731	0.69	1170	29908			1.02	955	24412	1042	26636
	129	66	2.57	986	25204	0.91	1298	33180			1.22	959	24514	1104	28221
	130	76	5.22	385	9842	1.69	622	15900			2.24	667	17050	648	16564
	131	68	2.69	735	18788										
	132	72	3.72	575	14698										
	133	69	3.04	853	21805										
	134	65	2.64	993	25383										
	135	70	3.03	721	18430										
	136	66	2.99	758	19376										
	137	78	4.61	473	12091										
	138	72	3.50	1018	26022										
	139	73	3.44	953	24361										
	140	74	3.80	676	17280										
	141	81	4.88	654	16718										
	142	72	3.74	791	20220										
	143	69	3.13	657	16794										
	144	69	3.16	411	10506										
	145	75	4.09	905	23134										
	146	71	3.28	665	16999										
	147	69	3.17	874	22342										
	148	65	2.61	715	18277										
	149	71	3.15	490	12526										
	150	71	3.27	758	19376										
	151	71	3.29	763	19504										
	152	73	3.94	434	11094										
	153	71	3.30	412	10532										
	154	68	2.93	641	16385										
	155	77	4.45	437	11171										
	156	68	2.78	1206	30828										
	157	77	4.31	541	13829										
	158	67	2.69	773	19760										
	159	70	3.16	801	20475										
	160	70	3.34	716	18303										
	161	69	3.02	1346	34407										
	162	62	2.19	824	21063										
	163	71	3.39	733	18737										
	164	68	2.80	834	21319										

Table A.3.18. Mean mercury content of yellow perch from FM82-1, a limnocorral located over a flooded moss/sedge/grass bottom, 1982. (For sample sizes, see Table A.3.17).

Date(s) sampled	Fish no.	Whole fish				Muscle tissue					Carcass			Combined muscle/carcass	
		Ttl lgth (mm)	Rnd wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	Total Hg (μg/g)	Specific activity (dpm/μg)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)
July 3,5	Time 0	56	1.43			0.53			0.043						
July 19,21	101-106	62	2.17			0.67	403	10302	0.060	169157	1.06	776	19837	635	16219
Aug 2	107-112	64	2.41			0.92	530	13557	0.055	246019	1.10	684	17493	616	15742
Aug 16	113-118	71	3.33			1.25	580	14818	0.051	287526	1.45	675	17250	630	16104
Aug 30	119-164	71	3.44	736	18807	1.33	788	20141	0.028	691934	1.68	723	18486	751	19195

Table A.3.19. Mercury content of yellow perch from LM82-3, a sediment-bottomed limnocorral with added (low) moss-peat, 1982.

Date(s) sampled	Fish no.	Whole fish				Muscle tissue					Carcass			Combined muscle/carcass	
		Ttl lgth (mm)	Rnd wt (g)	203Hg (cpm/g)	203Hg (dpm/g)	Wt (g)	203Hg (cpm/g)	203Hg (dpm/g)	Total Hg (µg/g)	Specific activity (dpm/µg)	Wt (g)	203Hg (cpm/g)	203Hg (dpm/g)	203Hg (cpm/g)	203Hg (dpm/g)
July 19,21	301	59	1.94			0.45	920	23517	0.051	461118	0.89	1452	37117	1273	32541
	302	66	2.59			0.72	427	10915	0.064	170547	1.33	854	21830	704	17996
	303	56	1.66			0.52	783	20015	0.038	526711	0.78	1270	32464	1075	27480
	304	62	2.07			0.62	508	12986	0.072	180361	1.00	1136	29039	896	22904
	305	55	1.57			0.53	748	19121	0.051	374922	0.69	1321	33768	1072	27403
	306	62	2.18			0.74	804	20552	0.053	387774	1.04	1083	27684	967	24719
Aug 2	307	70	3.30			1.29	696	17791	0.045	395356	1.50	971	24821	844	21575
	308	63	2.40			0.89	2546	65082	0.066	986091	1.11	2698	68967	2630	67229
	309	67	2.85			1.14	1387	35455	0.059	600932	1.40	1638	41871	1525	38983
	310	65	2.49			0.98	1815	46396	0.061	760590	1.14	1561	39903	1678	42894
	311	64	2.48			0.94	2147	54882	0.055	997855	1.18	2049	52377	2092	53476
	312	68	2.96			1.05	2164	55317	0.078	709192	1.35	2020	51636	2083	53246
Aug 16	313	68	2.67			0.93	3548	90695	0.072	1259653	1.35	2366	60481	2848	72802
	314	67	2.95			1.06	2029	51866	0.047	1103532	1.36	1577	40312	1775	45373
	315	70	3.11			1.17	4107	104985	0.072	1458125	1.33	2921	74668	3476	88855
	316	76	4.22			1.67	2774	70910	0.057	1244035	1.90	2150	54959	2442	62423
	317	72	3.19			1.19	3258	83282	0.072	1156694	1.47	1986	50767	2555	65312
	318	74	3.72			1.45	3229	82541	0.064	1289703	1.69	2287	58461	2722	69581
Aug 30	319	86	3.93			1.50	4372	111759	0.044	2539977	2.02	3374	86247	3799	97111
	320	76	3.62			1.47	5591	142919	0.045	3175978	1.49	3455	88318	4516	115440
	321	78	4.67			1.80	4012	102556	0.035	2930171	2.25	2537	64852	3193	81621
	322	69	3.03			1.17	3937	100639	0.090	1118211	1.20	2219	56723	3067	78400
	323	71	3.36			1.33	3883	99259	0.038	2612079	1.55	2515	64289	3147	80445
	324	72	3.57			1.44	8988	229755	0.110	2088682	1.72	5248	134151	6952	177710
	325	62	2.32	3059	78195	0.79	4963	126866							
	326	62	2.36	3488	89162	0.82	5502	140644							
	327	63	2.37	3478	88906	0.93	5392	137832							
	328	80	4.40	2784	71166	1.46	4394	112321							
	329	75	4.41	1982	50665	1.50	2970	75920							
	330	71	3.36	2879	73594	1.26	4351	111222							
	331	68	3.16	2546	65082	0.92	4476	114417			1.50	2847	72776	3466	88599
	332	74	4.15	2859	73083	1.35	4752	121472			1.93	3371	86171	3939	100690
	333	71	3.42	3926	100358	1.21	6290	160787			1.55	4021	102786	5016	128221
	334	73	3.79	2743	70118	1.37	4275	109279			1.66	2752	70348	3441	87960
	335	71	3.66	2369	60557	1.35	4303	109995			1.62	2620	66973	3385	86529
	336	71	3.55	3472	88753	1.30	6266	160174			1.56	3368	86094	4685	119760
	337	67	2.86	4716	120552										
	338	69	3.08	1560	39877										
	339	71	4.00	4845	123850										
	340	68	3.48	4258	108845										
	341	69	3.24	2846	72751										
	342	77	4.18	2408	61554										
	343	69	3.40	3095	79116										
	344	75	3.75	2524	64519										
	345	71	3.39	3223	82388										
	346	69	3.39	2216	56646										
	347	67	3.17	2795	71447										
	348	71	3.40	3229	82541										
	349	77	4.73	2664	68098										
	350	73	3.90	3453	88267										
	351	71	3.93	5825	148901										
	352	68	2.78	1959	50077										
	353	65	2.32	3001	76713										
	354	69	3.01	3264	83436										
	355	73	4.21	883	22572										
	356	67	3.17	2922	74693										
	357	70	3.30	5347	136682										
	358	66	2.95	5560	142127										
	359	71	3.48	2566	65593										
	360	67	3.30	3191	81570										
	361	66	2.39	2437	62296										
	362	76	3.92	1661	42459										
	363	75	4.05	1469	37551										
	364	68	3.18	3154	80624										
	365	65	2.81	2879	73594										
	366	69	2.91	2368	60532										
	367	65	2.54	2207	56416										
	368	66	2.61	2802	71626										
	369	71	3.58	1666	42587										
	370	68	2.70	2676	68405										
	371	72	3.57	3275	83717										
	372	85	5.94	3140	80266										
	373	76	3.84	2632	67280										
	374	70	3.05	2655	67868										
	375	74	3.98	1825	46651										
	376	65	2.70	1926	49233										
	377	72	3.73	2495	63778										
	378	73	3.78	2525	64545										
	379	75	3.94	2347	59995										
	380	70	2.95	3146	80419										

These fish (#325-#330) were dissected for muscle, skin, gut, liver, gills and carcass. (See Table A.3.31 for complete mercury data)

Table A.3.20. Mean mercury content of yellow perch from LM82-3, a sediment-bottomed limnocorral with added (low) moss-peat, 1982. (For sample sizes, see Table A.3.19).

Date(s) sampled	Fish no.	Whole fish				Muscle tissue					Carcass			Combined muscle/carcass	
		Ttl lgth (mm)	Rnd wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	Total Hg (μg/g)	Specific activity (dpm/μg)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)
July 3,5	Time 0	56	1.43			0.53			0.043						
July 19,21	301-306	60	2.00			0.60	698	17851	0.055	350239	0.96	1186	30317	998	25507
Aug 2	307-312	66	2.75			1.05	1793	45821	0.061	741669	1.28	1823	46596	1809	46234
Aug 16	313-318	71	3.31			1.25	3158	80713	0.064	1251957	1.52	2215	56608	2636	67391
Aug 30	319-380	71	3.45	2915	74505	1.28	4929	125990	0.040	2410850	1.67	3194	81644	4051	103541

Table A.3.21. Mercury content of yellow perch from HM82-4, a sediment-bottomed limnocorral with added (high) moss-peat 1982.

		Whole fish				Muscle tissue					Carcass			Combined muscle/carcass	
Date(s) sampled	Fish no.	Ttl lgth (mm)	Rnd wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	Total Hg (µg/g)	Specific activity (dpm/µg)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)
July 19,21	401	58	1.55			0.52	604	15440	0.047	328511	0.77	1266	32362	999	25537
	402	65	2.48			0.77	639	16334	0.055	296982	1.17	1064	27198	895	22878
	403	52	1.14			0.26	837	21396	0.049	436653	0.60	1685	43073	1429	36529
	404	61	1.77			0.55	699	17868	0.055	324873	0.89	1510	38599	1200	30675
	405	64	2.33			0.87	472	12065	0.057	211667	1.04	958	24489	737	18839
	406	60	1.88			0.61	563	14392	0.059	243932	0.93	1149	29371	917	23441
Aug 2	407	61	2.31			0.86	2562	65491	0.072	909597	0.98	2261	57797	2402	61401
	408	68	2.88			1.21	2324	59407	0.074	802797	1.31	2320	59305	2322	59356
	409	60	2.52			1.04	2556	65337	0.068	960838	1.03	2535	64801	2546	65082
	410	63	2.27			0.87	2794	71421	0.081	881741	1.09	2680	68507	2731	69811
	411	63	2.23			0.78	2786	71217	0.078	913038	1.02	2274	58129	2496	63804
	412	70	3.14			1.24	2249	57490	0.068	845441	1.43	1969	50332	2099	53655
Aug 16	413	69	3.99			1.51	2748	70245	0.074	949257	1.79	1927	49259	2303	58870
	414	69	3.32			1.42	5290	135225	0.120	1126875	1.30	3724	95194	4542	116104
	415	85	6.86			2.77	4929	125997	0.120	1049975	2.89	3232	82618	4063	103860
	416	70	3.29			1.23	3705	94709	0.072	1315403	1.36	2513	64238	3079	78707
	417	69	3.54			1.41	4628	118303	0.100	1183030	1.55	3369	86120	3969	101457
	418	74	4.34			1.72	4544	116155	0.110	1055955	1.85	3331	85148	3915	100077
Aug 30	419	79	5.04			2.09	5939	151815	0.084	1807321	2.08	3704	94683	4824	123313
	420	76	4.70			1.99	6378	163037	0.084	1940917	2.03	3580	91513	4965	126917
	421	73	3.72			1.49	5954	152198	0.092	1654326	1.68	3431	87704	4617	118021
	422	73	3.24			1.31	5088	130061	0.089	1461360	1.39	3113	79576	4071	104064
	423	76	4.08			1.75	6148	157157	0.086	1827407	1.77	3826	97802	4980	127301
	424	80	5.59			2.27	5305	135608	0.084	1614381	2.38	3170	81033	4212	107669
	425	71	4.11	3672	93865	1.40	5949	152071			1.92	3663	93635	4627	118277
	426	75	4.42	3238	82771	1.62	5148	131595			1.93	3011	76968	3986	101892
	427	73	4.17	3447	88113	1.45	6243	159586			1.86	3836	98057	4890	125000
	428	69	3.38	4153	106161	1.32	6277	160455			1.47	4228	108078	5197	132848
	429	79	5.44	3579	91488	1.93	6142	157004			2.26	3623	92612	4783	122265
	430	72	3.84	4595	117459	1.32	7533	192561			1.60	4530	115798	5888	150511
	431	69	3.29	2233	57081										
	432	77	4.70	2625	67101										
	433	89	7.20	1464	37423										
	434	71	3.70	3901	99719										
	435	82	5.92	3593	91846										
	436	75	4.26	4095	104678										
	437	84	6.02	4250	108640										
	438	89	7.33	3902	99744										
	439	79	5.37	1940	49591										
	440	82	5.89	2016	51534										
	441	70	3.38	1911	48850										
	442	72	3.98	2170	55470										
	443	80	5.34	2314	59151										
	444	74	4.29	4367	111631										
	445	72	3.88	3765	96242										
	446	81	5.68	3428	87628										
	447	84	6.33	3448	88139										
	448	81	5.86	3289	84075										
	449	73	4.53	3207	81979										
	450	70	3.90	2700	69018										
	451	79	5.40	3273	83666										
	452	84	6.24	3143	80343										
	453	74	4.76	2253	57592										
	454	67	3.24	2674	68354										
	455	85	6.71	1871	47827										
	456	79	5.26	1649	42152										
	457	72	3.62	1991	50895										
	458	77	4.64	2945	75281										
	459	78	4.86	3675	93942										
	460	69	3.45	2903	74208										
	461	81	5.77	6230	159254										
	462	80	5.16	3241	82848										

Table A.3.22. Mean mercury content of yellow perch from HM82-4,a sediment-bottomed limnocorral with added (high) moss-peat, 1982. (For sample sizes, see Table A.3.21).

		Whole fish				Muscle tissue					Carcass			Combined muscle/carcass	
Date(s) sampled	Fish no.	Ttl lgth (mm)	Rnd wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	Total Hg (μg/g)	Specific activity (dpm/μg)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)
July 3,5	Time 0	56	1.43			0.53			0.043						
July 19,21	401-406	60	1.86			0.60	636	16249	0.054	307103	0.90	1272	32515	1030	26317
Aug 2	407-412	64	2.56			1.00	2545	65061	0.074	885575	1.14	2340	59812	2433	62185
Aug 16	413-418	73	4.22			1.68	4307	110106	0.099	1113416	1.79	3016	77096	3645	93179
Aug 30	419-462	77	4.81	3136	80152	1.66	6009	153596	0.087	1717619	1.86	3643	93122	4753	121507

Table A.3.23. Mercury content of yellow perch from OC82-5, a sediment-bottomed control limnocorral, used for the second consecutive year, 1982.

Date(s) sampled	Fish no.	Whole fish				Muscle tissue					Carcass			Combined muscle/carcass	
		Ttl lgth (mm)	Rnd wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	Total Hg (μg/g)	Specific activity (dpm/μg)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)
July 19,21	501	60	2.24			0.71	4	102	0.083	1229	1.04	10	256	8	204
	502	66	2.91			1.05	109	2786	0.059	47220	1.25	160	4090	137	3502
	503	58	1.65			0.50	26	665	0.120	5542	0.73	32	818	30	767
	504	57	1.47			0.47	207	5291	0.120	44092	0.68	333	8512	282	7209
	505	62	1.98			0.68	154	3937	0.093	42333	0.87	233	5956	198	5061
	506	69	3.21			1.14	11	281	0.066	4258	1.45	29	741	21	537
Aug 2	507	61	2.11			0.87	38	971	0.040	24275	0.91	84	2147	62	1585
	508	65	2.20			0.78	64	1636	0.042	38952	1.17	126	3221	101	2582
	509	65	2.41			0.85	256	6544	0.049	133551	1.22	297	7592	280	7157
	510	72	3.49			1.30	41	1048	0.057	18386	1.76	81	2071	64	1636
Aug 16	511	59	2.10			0.76	656	16769	0.045	372644	0.93	568	14519	608	15542
	512	65	3.09			1.17	645	16488	0.076	216947	1.39	482	12321	556	14213
	513	71	3.69			1.30	490	12526	0.061	205344	1.67	488	12474	489	12500
	514	73	4.27			1.58	1027	26253	0.068	386074	1.83	676	17280	839	21447
	515	72	3.97			1.38	549	14034	0.053	264792	1.76	386	9867	458	11708
	516	73	4.03			1.57	667	17050	0.059	288983	1.79	414	10583	532	13599
Aug 30	517	83	6.32			2.58	1532	39162	0.034	1151824	2.79	1065	27224	1289	32950
	518	74	4.52			1.74	695	17766	0.030	592200	1.95	613	15670	652	16667
	519	74	4.16			1.65	1002	25613	0.050	512260	1.86	771	19709	880	22495
	520	74	4.17			1.63	1218	31135	0.034	915735	1.88	834	21319	1012	25869
	521	74	3.95			1.64	415	10608	0.038	279158	1.64	401	10251	408	10429
	522	75	4.52			1.87	929	23747	0.019	1249842	1.83	692	17689	812	20757
	523	75	4.57	753	19248	1.52	1404	35890			1.92	786	20092	1059	27071
	524	70	3.75	871	22265	1.22	1348	34458			1.58	899	22981	1095	27991
	525	67	3.12	798	20399	1.06	1327	33921			1.41	704	17996	971	24821
	526	66	2.69	494	12628	0.83	668	17076			1.30	432	11043	524	13395
	527	64	2.82	1232	31493	1.01	1732	44274			1.27	1005	25690	1327	33921
	528	67	3.28	676	17280	1.19	911	23287			1.45	654	16718	770	19683
	529	70	3.75	1191	30445										

Table A.3.24. Mean mercury content of yellow perch from OC82-5, a sediment-bottomed control limnocorral, used for the second consecutive year, 1982. (For sample sizes, see Table A.3.23).

Date(s) sampled	Fish no.	Whole fish				Muscle tissue					Carcass			Combined muscle/carcass	
		Ttl lgth (mm)	Rnd wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	Total Hg (μg/g)	Specific activity (dpm/μg)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)
July 3,5	Time 0	56	1.43			0.53			0.043						
July 19,21	501-506	62	2.24			0.76	85	2177	0.090	24112	1.00	133	3396	113	2880
Aug 2	507-510	66	2.55			0.95	100	2550	0.047	53791	1.27	147	3758	127	3240
Aug 16	511-516	69	3.53			1.29	672	17187	0.060	289131	1.56	502	12841	580	14835
Aug 30	517-529	72	3.97	859	21965	1.50	1098	28078	0.034	783503	1.74	738	18865	900	23004

Table A.3.25. Mercury content of yellow perch from C82-6, a sediment-bottomed control limnocorral, 1982.

		Whole fish				Muscle tissue					Carcass			Combined muscle/carcass	
Date(s) sampled	Fish no.	Ttl lgth (mm)	Rnd wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	Total Hg (μg/g)	Specific activity (dpm/μg)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)
July 19,21	601	53	1.46			0.49	765	19555	0.066	296288	0.69	1406	35941	1140	29141
	602	59	1.83			0.65	530	13548	0.070	193543	0.87	1342	34305	995	25435
	603	58	1.85			0.67	741	18942	0.055	344400	0.87	1349	34484	1084	27710
	604	55	1.31			0.48	648	16564	0.055	301164	0.60	1532	39162	1139	29116
	605	57	2.03			0.66	617	15772	0.042	375524	0.95	1430	36554	1097	28042
	606	61	2.12			0.76	472	12065	0.045	268111	0.93	1190	30419	867	22163
Aug 2	607	60	1.97			0.61	428	10941	0.036	303917	1.00	1115	28502	855	21856
	608	65	2.27			0.85	386	9867	0.055	179400	1.18	1482	37883	1023	26150
	609	65	2.74			1.02	282	7209	0.061	118180	1.33	736	18814	539	13778
	610	72	3.34			1.28	315	8052	0.042	191714	1.55	847	21651	606	15491
	611	65	2.71			0.97	186	4755	0.038	125132	1.27	683	17459	468	11963
	612	66	2.80			1.08	411	10506	0.047	223532	1.32	889	22725	674	17229
Aug 16	613	67	2.56			0.83	447	11426	0.074	154405	1.28	697	17817	599	15312
	614	70	3.76			1.32	797	20373	0.076	268066	1.57	864	22086	833	21293
	615	73	3.99			1.53	494	12628	0.051	247608	1.56	699	17868	597	15261
	616	74	3.63			1.36	351	8972	0.057	157404	1.79	687	17561	542	13855
	617	70	2.76			1.06	586	14980	0.066	226970	1.21	855	21856	729	18635
	618	67	2.77			1.02	454	11605	0.055	211000	1.23	884	22597	689	17612
Aug 30	619	74	4.12			1.68	559	14289	0.023	621261	1.78	645	16488	603	15414
	620	83	5.31			2.13	365	9330	0.032	291563	2.58	501	12807	439	11222
	621	84	5.99			2.47	427	10915	0.035	311857	2.66	592	15133	513	13113
	622	74	3.56			1.38	639	16334	0.024	680583	1.61	779	19913	714	18252
	623	66	3.07			1.14	666	17025	0.063	270238	1.39	662	16922	664	16973
	624	76	4.18			1.65	507	12960	0.028	462857	1.84	659	16846	587	15005
	625	79	4.73	569	14545	1.67	538	13753							
	626	70	3.52	668	17076	1.29	632	16155							
	627	60	1.70	922	23569	0.53	829	21191							
	628	70	3.12	753	19248	1.20	712	18200							
	629	84	6.37	485	12398	2.34	361	9228							
	630	80	5.09	647	16539	1.85	342	8742							
	631	70	3.16	1152	29448	1.00	943	24105			1.50	1141	29167	1062	27147
	632	69	3.29	547	13983	1.10	450	11503			1.41	676	17280	577	14749
	633	72	3.72	758	19376	1.15	582	14877			1.73	811	20731	720	18405
	634	73	3.75	823	21038	1.28	727	18584			1.61	927	23696	838	21421
	635	76	4.79	479	12244	1.55	451	11529			1.91	622	15900	545	13931
	636	70	3.33	781	19964	1.14	519	13267			1.49	535	13676	528	13497
	637	75	4.11	772	19734										
	638	64	2.56	764	19530										
	639	67	2.82	659	16846										
	640	66	3.08	836	21370										
	641	67	2.65	504	12883										
	642	67	2.70	953	24361										
	643	69	3.21	838	21421										
	644	65	2.62	958	24489										
	645	64	2.78	662	16922										
	646	70	3.73	662	16922										
	647	70	3.47	750	19172										
	648	66	2.75	726	18558										
	649	74	4.31	704	17996										
	650	66	3.16	816	20859										
	651	78	5.72	711	18175										
	652	66	3.06	709	18124										
	653	73	3.86	740	18916										
	654	71	3.64	741	18942										
	655	75	4.16	524	13395										
	656	66	2.71	1025	26201										
	657	62	2.38	781	19964										
	658	58	2.05	717	18328										
	659	67	3.17	806	20603										
	660	67	3.17	888	22699										
	661	76	4.76	483	12347										
	662	67	2.83	698	17843										
	663	63	2.62	817	20884										
	664	70	3.27	672	17178										
	665	67	2.98	846	21626										
	666	66	2.79	664	16973										
	667	67	2.96	716	18303										
	668	68	3.14	711	18175										
	669	70	3.57	729	18635										
	670	69	2.99	806	20603										
	671	72	4.11	640	16360										
	672	66	2.58	754	19274										
	673	70	3.43	664	16973										
	674	67	2.74	647	16539										
	675	77	4.61	576	14724										
	676	66	2.69	964	24642										
	677	72	3.50	659	16846										
	678	78	5.00	540	13804										
	679	72	3.28	662	16922										

These fish (#625-#630) were dissected for muscle, skin, gut, liver, gills and carcass. (See Table A.3.31 for complete mercury data)

Table A.3.26. Mean mercury content of yellow perch from C82-6, a sediment-bottomed control limnocorral, 1982. (For sample sizes, see Table A.3.25).

		Whole fish				Muscle tissue					Carcass			Combined muscle/carcass	
Date(s) sampled	Fish no.	Ttl lgth (mm)	Rnd wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	Total Hg (µg/g)	Specific activity (dpm/µg)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)
July 3,5	Time 0	56	1.43			0.53			0.043						
July 19,21	601-606	57	1.77			0.62	629	16074	0.056	296505	0.82	1375	35144	1054	26935
Aug 2	607-612	66	2.64			0.97	335	8555	0.047	190313	1.28	959	24506	694	17745
Aug 16	613-618	70	3.25			1.19	522	13331	0.063	210909	1.44	781	19964	665	16995
Aug 30	619-679	70	3.52	729	18627	1.48	569	14555	0.023	439727	1.79	713	18213	649	16594

Table A.3.27. Mercury content of yellow perch from HS82-7, a sediment-bottomed limnocorral with added (high) bank clay, 1982.

		Whole fish				Muscle tissue					Carcass			Combined muscle/carcass	
Date(s) sampled	Fish no.	Ttl lgth (mm)	Rnd Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	Total Hg (µg/g)	Specific activity (dpm/µg)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)
July 19,21	701	67	2.68			1.04	288	7362	0.059	124780	1.29	566	14468	442	11299
	702	59	1.51			0.50	673	17203	0.064	268797	0.73	885	22623	799	20424
	703	65	2.47			0.95	412	10532	0.064	164563	1.15	713	18226	577	14749
	704	59	1.86			0.67	458	11708	0.047	249106	0.91	839	21447	677	17306
	705	64	2.56			0.88	601	15363	0.053	289868	1.12	967	24719	806	20603
	706	64	2.29			0.85	371	9484	0.038	249579	1.08	742	18967	579	14801
Aug 2	707	62	2.73			1.04	1518	38804	0.055	705527	1.20	1335	34126	1420	36299
	708	67	2.53			0.89	776	19836	0.040	495900	1.23	885	22623	839	21447
	709	57	1.85			0.63	1479	37807	0.057	663281	0.87	1356	34663	1408	35992
	710	62	2.08			0.72	1679	42919	0.057	752965	0.99	1608	41104	1638	41871
	711	71	3.37			1.25	600	15337	0.040	383425	1.53	683	17459	646	16513
	712	65	2.36			0.80	1388	35481	0.042	844786	1.18	1495	38216	1452	37117
Aug 16	713	76	3.98			1.36	2558	65389	0.074	883635	1.92	1902	48620	2174	55573
	714	62	1.80			0.62	1581	40414	0.047	859872	0.93	1111	28400	1299	33206
	715	57	1.60			0.49	4357	111375	0.059	1887712	0.74	2401	61375	3180	81288
	716	70	2.77			1.00	947	24208	0.053	456755	1.36	743	18993	829	21191
	717	68	2.69			1.00	3243	82899	0.055	1507255	1.28	2263	57848	2693	68839
	718	62	2.06			0.76	3322	84918	0.066	1286636	0.98	2116	54090	2643	67561
Aug 30	719	72	3.47			1.38	2659	67970	0.087	781264	1.60	1654	42280	2119	54167
	720	74	3.75			1.32	3308	84560	0.087	971954	1.74	2019	51610	2575	65823
	721	78	4.36			1.68	1672	42740	0.083	514940	1.99	986	25204	1300	33231
	722	65	2.56			1.00	3324	84969	0.100	849690	1.21	2390	61094	2813	71907
	723	72	3.02			1.03	2967	75844	0.100	758440	1.62	1711	43737	2199	56212
	724	65	2.42			0.92	3877	99105	0.110	900955	1.04	2015	51508	2889	73850
	725	69	3.16	2266	57924	0.98	3480	88957			1.42	2438	62321	2863	73185
	726	62	2.45	2813	71907	0.73	4503	115107			1.16	2853	72929	3490	89213
	727	64	2.76	2138	54652	0.91	3504	89571			1.35	2028	51840	2622	67025
	728	82	4.93	1767	45169	1.44	2793	71396			2.38	1846	47188	2203	56314
	729	70	3.38	1556	39775	1.15	2970	75920			1.52	1493	38165	2129	54422
	730	66	2.77	2826	72239	0.97	3832	97955			1.33	2708	69223	3182	81339
	731	63	2.31	2209	56467										
	732	66	2.55	2562	65491										
	733	71	3.42	1880	48057										
	734	71	3.50	1085	27735										
	735	65	2.44	2034	51994										
	736	66	2.48	2359	60302										
	737	67	2.76	2569	65670										
	738	66	2.80	2477	63318										
	739	60	2.08	2155	55087										
	740	71	2.97	2505	64034										
	741	60	2.20	2374	60685										
	742	71	3.11	1761	45015										
	743	70	3.16	1904	48671										
	744	65	2.51	1899	48543										
	745	62	2.36	1942	49642										
	746	67	2.76	3217	82234										
	747	61	2.25	2157	55138										
	748	57	1.85	2607	66641										
	749	77	4.51	1456	37219										
	750	61	2.49	2790	71319										
	751	67	3.08	2676	68405										
	752	66	2.52	2883	73696										
	753	66	2.70	2486	63548										
	754	77	4.58	755	19300										
	755	63	2.26	987	25230										
	756	63	2.44	2616	66871										
	757	66	2.66	1897	48492										
	758	76	4.48	804	20552										
	759	67	2.87	3090	78988										
	760	71	3.95	763	19504										
	761	67	2.84	2260	57771										
	762	66	2.77	2600	66462										
	763	67	3.13	1674	42791										
	764	64	2.55	2810	71830										
	765	67	2.83	2575	65823										
	766	64	2.79	2640	67485										
	767	72	3.65	1379	35251										
	768	70	3.21	990	25307										
	769	66	2.59	3139	80240										
	770	69	2.74	2192	56033										
	771	68	2.56	1945	49719										

Table A.3.28. Mean mercury content of yellow perch from HS82-7, a sediment-bottomed limnocorral with added (high) bank clay, 1982. (For sample sizes, see Table A.3.27).

		Whole fish				Muscle tissue					Carcass			Combined muscle/carcass	
Date(s) sampled	Fish no.	Ttl lgth (mm)	Rnd wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	Total Hg (μg/g)	Specific activity (dpm/μg)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)
July 3,5	Time 0	56	1.43			0.53			0.043						
July 19,21	701-706	63	2.23			0.82	467	11942	0.054	224449	1.05	785	20075	647	16530
Aug 2	707-712	64	2.49			0.89	1240	31697	0.049	640981	1.17	1227	31365	1234	31540
Aug 16	713-718	66	2.48			0.87	2668	68201	0.059	1146978	1.2	1756	44888	2136	54610
Aug 30	719-771	68	2.96	2138	54643	1.13	3241	82841	0.095	796207	1.53	2012	51425	2532	64724

Table A.3.29. Mercury content of yellow perch from OM82-8, a sediment-bottomed limnocorral with added (low) moss from 1981 and used for the second consecutive year, 1982.

		Whole fish				Muscle tissue					Carcass			Combined muscle/carcass	
Date(s) sampled	Fish no.	Ttl lgth (mm)	Rnd Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	Total Hg (μg/g)	Specific activity (dpm/μg)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)
July 19,21	801	63	2.22			0.76	654	16790	0.098	171327	1.14	979	25133	849	21796
	802	60	1.82			0.68	1011	25955	0.130	199654	0.91	1249	32065	1147	29446
	803	61	2.10			0.79	501	12862	0.085	151318	0.93	958	24594	748	19203
	804	65	2.27			0.74	563	14454	0.068	212559	1.09	784	20127	695	17842
	805	65	2.32			0.87	680	17457	0.087	200655	1.11	1047	26879	886	22746
	806	67	2.22			0.92	576	14787	0.085	173965	1.01	750	19254	667	17124
Aug 2	807	66	2.48			0.81	1861	47777	0.140	341264	1.35	1876	48162	1870	48008
	808	72	3.16			1.18	1584	40665	0.130	312808	1.48	1342	34453	1449	37200
	809	68	3.11			1.20	1722	44208	0.130	340062	1.43	1659	42591	1688	43335
	810	56	1.80			0.57	1051	26982	0.076	355026	0.87	1118	28702	1091	28009
	811	66	2.59			0.97	1214	31167	0.110	283336	1.25	1198	30756	1205	30936
	812	71	3.78			1.42	1053	27033	0.093	290677	1.66	951	24415	998	25621
Aug 16	813	77	4.63			1.70	1601	41102	0.120	342517	2.07	1145	29395	1351	34684
	814	68	2.75			1.10	1864	47854	0.130	368108	1.23	1332	34196	1583	40640
	815	65	2.60			0.99	3647	93628	0.210	445848	1.20	2458	63103	2995	76890
	816	78	4.45			1.77	1114	28599	0.110	259991	1.90	978	25108	1044	26802
	817	71	3.42			1.32	2106	54067	0.150	360447	1.51	1594	40922	1833	47058
	818	76	4.40			1.74	2049	52603	0.130	404638	1.85	1582	40614	1808	46416
Aug 30	819	76	4.11			1.67	1322	33939	0.130	261069	1.78	1075	27598	1195	30679
	820	73	3.93			1.59	1921	49317	0.150	328780	1.72	1238	31783	1566	40203
	821	69	3.01			1.20	2700	69316	0.190	364821	1.29	1573	40383	2116	54323
	822	79	5.08			2.12	1577	40486	0.140	289186	2.08	1214	31167	1397	35865
	823	75	4.32			1.76	1692	43438	0.190	228621	1.93	1158	29729	1413	36275
	824	70	3.20			1.31	3201	82178	0.260	316069	1.37	1755	45055	2462	63206
	825	70	4.05	1459	37456	1.43	2282	58585			1.80	1281	32887	1724	44260
	826	69	3.17	1245	31962	1.18	1912	49086			1.36	1325	34016	1598	41025
	827	78	4.99	1817	46647	1.73	2869	73655			2.23	1857	47674	2299	59021
	828	71	3.32	1908	48983	1.22	2869	73655			1.55	1607	41256	2163	55530
	829	71	3.61	1522	39074	1.24	2418	62076			1.74	1692	43438	1994	51191
	830	70	3.50	2087	53579	1.23	3227	82846			1.61	2274	58380	2687	68982
	831	74	4.25	1443	37046										
	832	72	3.89	1612	41384										
	833	75	4.42	1422	36506										
	834	74	3.77	1374	35274										
	835	72	3.85	1306	33528										
	836	81	6.00	1211	31090										
	837	66	2.93	2256	57917										
	838	82	6.41	1362	34966										
	839	71	3.77	1873	48085										
	840	64	2.59	2994	76864										
	841	73	4.27	1282	32912										
	842	72	3.20	3109	79816										
	843	76	4.56	1383	35505										
	844	75	4.55												
	845	65	2.78	3046	78199										
	846	64	2.82	2260	58020										
	847	77	5.44	811	20820										
	848	72	4.30	4045	103846										
	849	68	3.14	2415	61999										
	850	77	4.40	1361	34940										
	851	77	5.19	1428	36661										
	852	83	6.13	1117	28676										
	853	78	5.19	638	16379										
	854	69	3.27	2168	55658										
	855	70	3.87	1154	29626										
	856	73	4.38	3017	77454										
	857	74	4.06	1551	39818										
	858	73	3.96	1652	42411										
	859	81	5.49	1009	25904										
	860	82	5.77	1145	29395										
	861	65	2.56	2245	57635										
	862	66	2.79	2749	70574										
	863	84	5.94	1017	26109										
	864	74	4.53	1378	35377										
	865	76	5.06	1893	48598										

Table A.3.30. Mean mercury content of yellow perch from OM82-8, a sediment-bottomed limnocorral with added (low) moss from 1981 and used for the second consecutive year, 1982. (For sample sizes, see Table A.3.29).

		Whole fish				Muscle tissue					Carcass			Combined muscle/carcass	
Date(s) sampled	Fish no.	Ttl lgth (mm)	Rnd wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	Total Hg (µg/g)	Specific activity (dpm/µg)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)
July 3,5	Time 0	56	1.43			0.53			0.043						
July 19,21	801-806	64	2.16			0.79	664	17051	0.092	184913	1.03	961	24675	832	21360
Aug 2	807-812	67	2.82			1.03	1414	36305	0.113	320529	1.34	1357	34847	1384	35518
Aug 16	813-818	73	3.71			1.44	2064	52976	0.142	363592	1.63	1515	38890	1769	45415
Aug 30	819-865	73	4.17	1769	45417	1.47	2333	59881	0.177	298091	1.71	1504	38614	1885	48380

Table A.3.31. Distribution of ^{203}Hg activity (cpm/g & dpm/g) and total ^{203}Hg burden (cpm & dpm) in yellow perch organs and tissues at the end (Aug 30) of the limnocorral experiment, 1982.

Limnocorral treatment (designator)	Fish no.	Whole fish					Muscle					Skin					Liver				
		Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	^{203}Hg (cpm)	^{203}Hg (dpm)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	^{203}Hg (cpm)	^{203}Hg (dpm)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	^{203}Hg (cpm)	^{203}Hg (dpm)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	^{203}Hg (cpm)	^{203}Hg (dpm)
Control (C82-6)	625	4.73	569	14545	2691	68788	1.67	538	13753	898	22955	0.31	480	12270	149	3809	0.02	3998	102198	80	2045
	626	3.52	668	17076	2351	60097	1.29	632	16155	815	20833	0.19	539	13778	102	2607	0.03	3684	94172	111	2837
	627	1.70	922	23569	1567	40056	0.53	829	21191	439	11222	0.09	700	17894	63	1610	0.01	4399	112449	44	1125
	628	3.12	753	19248	2349	60046	1.20	712	18200	854	21830	0.19	613	15670	116	2965	0.04	4441	113522	178	4550
	629	6.37	485	12398	3089	78962	2.34	361	9228	845	21600	0.36	357	9126	129	3298	0.09	3137	80189	282	7209
	630	5.09	647	16539	3293	84177	1.85	342	8742	633	16181	0.28	418	10685	117	2991	0.04	5249	134177	210	5368
	Means:	4.09	674	17229	2557	65354	1.48	569	14545	747	19104	0.24	518	13237	113	2880	0.04	4151	106118	151	3856
Low Moss (LM82-3)	325	2.32	3059	78195	7097	181416	0.79	4963	126866	3921	100230	0.10	3121	79780	312	7975	0.01	6944	177505	69	1764
	326	2.36	3488	89162	8232	210429	0.82	5502	140644	4512	115337	0.14	3005	76815	421	10762	0.01	8484	216871	85	2173
	327	2.37	3478	88906	8243	210711	0.93	5392	137832	5015	128195	0.14	2644	67587	370	9458	0.03	6676	170654	200	5112
	328	4.40	2784	71166	12250	313139	1.46	4394	112321	6415	163983	0.37	2806	71728	1038	26534	0.02	5991	153144	120	3067
	329	4.41	1982	50665	8741	223441	1.50	2970	75920	4455	113880	0.33	1802	46063	595	15210	0.04	5932	151636	237	6058
	330	3.36	2879	73594	9673	247265	1.26	4351	111222	5482	140133	0.20	1995	50997	399	10199	0.02	5018	128272	100	2556
	Means:	3.20	2945	75281	9039	231067	1.13	4595	117468	4967	126960	0.21	2562	65495	523	13356	0.02	6508	166347	135	3455

Limnocorral treatment (designator)	Fish no.	Gut					Gill					Carcass					Tissue/organ totals				
		Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	^{203}Hg (cpm)	^{203}Hg (dpm)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	^{203}Hg (cpm)	^{203}Hg (dpm)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	^{203}Hg (cpm)	^{203}Hg (dpm)	Total[1] Wt (g)	Combined[2] ^{203}Hg (cpm/g)	Combined[2] ^{203}Hg (dpm/g)	Total[1] ^{203}Hg (cpm)	Total[1] ^{203}Hg (dpm)
Control (C82-6)	625	0.30	1780	45501	534	13650	0.14	1419	36273	199	5087	1.53	671	17152	1027	26253	3.97	727	18584	2887	73799
	626	0.26	1458	37270	379	9688	0.08	1304	33333	104	2658	1.16	712	18200	826	21115	3.01	777	19862	2337	59739
	627	0.18	1597	40823	287	7336	0.06	1748	44683	105	2684	0.57	1078	27556	614	15695	1.44	1079	27582	1552	39673
	628	0.17	1838	46984	312	7975	0.09	1582	40440	142	3630	1.03	853	21805	879	22469	2.72	912	23313	2481	63420
	629	0.71	1144	29243	812	20757	0.24	1038	26534	249	6365	1.84	497	12704	914	23364	5.58	579	14801	3231	82592
	630	0.36	3439	87909	1238	31646	0.22	1236	31595	272	6953	1.65	611	15619	1008	25767	4.40	790	20194	3478	88906
	Means:	0.33	1876	47955	594	15175	0.14	1388	35476	179	4563	1.30	737	18839	878	22444	3.52	811	20723	2661	68022
Low Moss (LM82-3)	325	0.09	4328	110634	390	9969	0.06	4518	115491	271	6927	0.75	3440	87935	2580	65951	1.80	4191	107132	7543	192817
	326	0.13	3098	79192	403	10302	0.06	4909	125486	295	7541	0.80	3898	99642	3118	79703	1.96	4507	115210	8834	225818
	327	0.12	4089	104525	491	12551	0.06	4511	115312	271	6927	0.80	3885	99310	3108	79448	2.08	4545	116181	9455	241692
	328	0.28	2705	69146	757	19351	0.14	3897	99617	546	13957	1.51	3358	85838	5071	129627	3.78	3690	94325	13947	356518
	329	0.21	2785	71191	585	14954	0.15	3473	88778	521	13318	1.50	2285	58410	3428	87628	3.73	2633	67306	9821	251048
	330	0.17	1577	40312	268	6851	0.11	3414	87270	376	9611	1.07	3047	77889	3260	83333	2.83	3493	89289	9885	252684
	Means:	0.17	3097	79167	482	12330	0.10	4120	105326	380	9714	1.07	3319	84837	3428	87615	2.70	3843	98241	9914	253430

[1] Sum of values for muscle, skin, liver, gut, gill and carcass.

[2] Mathematical combination of values for muscle, skin, liver, gut, gill and carcass.

Table A.3.32. "Time zero" data on yellow perch, 1983.

		Whole fish		Muscle	
Date(s) sampled	Fish no.	Total length (mm)	Round weight (g)	Wt (g)	Total Hg (μg/g)
July 8	1	73	3.91	1.48	0.120
	2	73	4.35	1.76	0.120
	3	73	3.70		
	4	79	4.61		
	5	64	2.12	0.81	0.140
	6	74	3.56		
	7	77	3.98		
	8	80	4.56		
	9	72	3.32		
	10	67	2.79	1.17	0.078
	11	74	3.90		
	12	70	2.98		
	13	72	3.11	1.28	0.074
	14	75	3.59		
	15	77	4.37		
	16	67	2.87	1.15	0.100
	17	70	2.94		
	18	78	5.16	1.87	0.088
	19	75	3.88		
	20	71	3.38		
	21	80	5.06		
July 10	22	71	3.47		
	23	64	2.25	0.82	0.130
	24	79	4.36		
	25	67	2.60	0.96	0.120
	26	62	1.92		
	27	60	1.58		
	28	76	3.84		
June 30	29	66	2.21	0.75	0.066
July 2	30	65	2.29		
July 10	31	79	4.69	2.05	0.052
	32	89	7.28	3.03	0.062
	33	75	4.10		

Table A.3.33. Mercury content of yellow perch from DFM83-1, a limnocorral located over a flooded, "disturbed" moss/sedge/grass bottom and used for the second consecutive year, 1983.

		Whole fish		Muscle tissue					Carcass			Combined muscle/carcass	
Date(s) sampled	Fish no.	Total length (mm)	Round wt (g)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	Total Hg (μg/g)	Specific activity (dpm/μg)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)
July 20,21	101	76	4.24	1.39	616	15746	0.100	157460	2.38	813	20782	740	18916
	102	76	3.51	1.18	403	10302	0.088	117068	1.90	709	18124	592	15133
	103	71	2.99	1.14	487	12449	0.081	153691	1.50	848	21677	692	17689
	104	73	3.44	1.35	654	16718	0.056	298536	1.71	1141	29167	926	23671
	105	73	3.45	1.21	504	12883	0.052	247750	1.70	769	19657	659	16846
	106	66	2.83	0.90	573	14647	0.120	122058	1.49	790	20194	708	18098
Aug 10	107	82	6.16	2.28	3076	78630	0.110	714818	2.76	2367	60506	2688	68712
	108	78	6.08	2.16	1557	39801	0.095	418958	2.80	1320	33742	1423	36375
	109	93	9.11	3.36	2064	52761	0.100	527610	4.30	1378	35225	1679	42919
	110	84	7.34	2.74	1551	39647	0.097	408732	3.17	1384	35378	1461	37347
	111	76	5.22	2.01	1866	47699	0.091	524165	2.42	1376	35174	1598	40849
	112	92	8.43	3.43	1288	32924	0.078	422103	3.36	997	25486	1144	29243
Aug 29	113	91	8.26	3.18	7526	192382	0.190	1012537	3.86	4924	125869	6099	155905
	114	80	5.58	2.04	6664	170348	0.170	1002047	2.43	3805	97265	5110	130624
	115	93	8.04	3.26	4815	123083	0.160	769269	3.52	3049	77940	3898	99642
	116	91	8.36	3.27	4657	119044	0.120	992033	3.86	3073	78553	3799	97111
	117	88	7.27	2.90	4963	126866	0.140	906186	3.05	3299	84330	4110	105061
	118	97	8.89	3.76	3133	80087	0.100	800870	3.95	2293	58615	2703	69095
	119	86	7.15	2.59	4955	126662	0.130	974323	3.20	3252	83129	4014	102607
	120	97	8.99	3.24	4805	122827	0.140	877336	4.05	3236	82720	3933	100537
	121	85	6.73	2.63	7665	195936	0.190	1031242	3.16	4962	126840	6190	158231
	122	90	8.64	3.32	4784	122290	0.130	940692	3.72	3193	81621	3943	100792

Table A.3.34. Mean mercury content of yellow perch from DFM83-1, a limnocorral located over a flooded, "disturbed" moss/sedge/grass bottom and used for the second consecutive year, 1983. (For sample sizes, see Table A.3.33).

		Whole fish		Muscle tissue					Carcass			Combined muscle/carcass	
Date(s) sampled	Fish no.	Total length (mm)	Round wt (g)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	Total Hg (μg/g)	Specific activity (dpm/μg)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)
June 30-July 10	Time 0	73	3.60	1.43			0.096						
July 20,21	101-106	73	3.41	1.20	540	13791	0.083	182761	1.78	845	21600	720	18392
Aug 10	107-112	84	7.06	2.66	1900	48577	0.095	502731	3.14	1470	37585	1666	42574
Aug 29	113-122	90	7.79	3.02	5397	137953	0.147	930654	3.48	3509	89688	4380	111961

Table A.3.35. Mercury content of yellow perch from SP83-2, a sediment-buttomed limnocorral with added black spruce boughs, 1983. (Shortly after the July 21 sample, the limnocorral went anoxic, killing all remaining fish).

Date(s) sampled	Fish no.	Whole fish		Muscle tissue					Carcass			Combined muscle/carcass	
		Total length (mm)	Round wt (g)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	Total Hg (μg/g)	Specific activity (dpm/μg)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)
July 20,21	201	75	3.87	1.42	774	19785	0.110	179864	2.05	1019	26048	919	23492
	202	80	5.41	1.98	521	13318	0.120	110983	2.76	685	17510	616	15746
	203	65	2.12	0.73	1336	34151	0.110	310464	1.11	1458	37270	1410	36043
	204	76	3.81	1.37	839	21447	0.076	282197	2.05	873	22316	859	21958
	205	73	2.97	1.16	286	7311	0.044	166159	1.43	474	12117	390	9969
	206	70	3.00	1.11	760	19427	0.120	161892	1.57	939	24003	865	22111
	207	79	3.70	1.62	323	8257	0.058	142362	1.65	386	9867	355	9075
	208	77	4.16	1.84	324	8282	0.062	133581	1.93	319	8154	321	8206
MEANS:													
June 30-July 10	Time 0	73	3.60	1.43			0.096						
July 20,21	201-208	74	4.00	1.40	645	16497	0.088	185938	1.82	769	19661	717	18325

Table A.3.36. Mercury content of yellow perch from LM83-3, a sediment-bottomed limnocorral with added (low) moss, 1983.

Date(s) sampled	Fish no.	Whole fish		Muscle tissue					Carcass			Combined muscle/carcass	
		Total length (mm)	Round wt (g)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	Total Hg (μg/g)	Specific activity (dpm/μg)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)
July 20,21	301	67	2.73	1.08	463	11835	0.100	118350	1.27	797	20373	644	16462
	302	65	2.47	1.01	526	13446	0.069	194870	1.16	803	20527	674	17229
	303	74	4.21	1.59	440	11247	0.087	129276	1.91	816	20859	645	16488
	304	76	4.14	1.60	439	11222	0.120	93517	1.91	826	21115	650	16616
	305	82	6.00	2.40	204	5215	0.094	55479	2.90	527	13471	381	9739
	306	84	5.76	2.15	206	5266	0.100	52660	3.05	505	12909	381	9739
Aug 10	307	87	6.43	2.72	2575	65823	0.130	506331	2.98	2686	68661	2633	67306
	308	80	5.07	1.82	2502	63957	0.130	491977	2.35	2452	62679	2474	63241
	309	85	6.46	2.39	1748	44683	0.120	372358	3.05	1945	49719	1858	47495
	310	80	5.30	1.91	1648	42127	0.100	421270	2.46	1866	47699	1771	45271
	311	85	6.00	2.41	3084	78834	0.160	492713	2.47	3126	79908	3105	79371
	312	94	8.53	3.30	1192	30470	0.072	423194	3.93	1409	36017	1310	33487
Aug 29	313	94	8.18	3.26	4826	123364	0.180	685356	3.50	3103	79320	3934	100562
	314	92	8.17	3.32	3549	90721	0.110	824736	3.64	2203	56314	2845	72725
	315	90	7.08	2.80	7947	203144	0.240	846433	3.00	5694	145552	6782	173364
	316	82	5.59	2.40	5572	142434	0.150	949560	2.47	3894	99540	4721	120680
	317	108	11.59	4.62	1355	34637	0.100	346370	5.46	1156	29550	1247	31876
	318	78	4.34	1.84	5321	136017	0.170	800100	1.92	3396	86810	4338	110890
	319	100	9.45	3.76	4233	108206	0.160	676288	4.23	2871	73390	3512	89775
	320	80	5.05	2.11	5130	131135	0.120	1092792	2.42	3553	90823	4288	109611
	321	94	7.97	3.10	4669	119351	0.170	702065	3.98	3100	79243	3787	96805
	322	93	8.66	3.38	2794	71421	0.120	595175	3.91	1913	48901	2321	59330
	323	86	6.97	2.85	3778	96575	0.140	689821	3.32	2411	61631	3042	77761
	324	94	8.96	3.64	3197	81723	0.120	681025	3.97	2591	66232	2881	73645
	325	83	6.07	2.28	5762	147290	0.160	920563	2.81	3849	98390	4706	120297
	326	81	5.18	2.03	11042	282260	0.260	1085615	2.26	6974	178272	8899	227480
	327	92	9.52	3.58	3896	99591	0.120	829925	4.03	2566	65593	3192	81595

Table A.3.37. Mean mercury content of yellow perch from LM83-3, a sediment-bottomed limnocorral with added (low) moss, 1983. (For sample sizes, see Table A.3.36).

Date(s) sampled	Fish no.	Whole fish		Muscle tissue					Carcass			Combined muscle/carcass	
		Total length (mm)	Round wt (g)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	Total Hg (μg/g)	Specific activity (dpm/μg)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)
June 30-July 10	Time 0	73	3.60	1.43			0.096						
July 20,21	301-306	75	4.22	1.64	380	9705	0.095	107359	2.03	712	18209	563	14379
Aug 10	307-312	85	6.30	2.43	2125	54316	0.119	451307	2.87	2247	57447	2192	56029
Aug 29	313-327	90	7.52	3.00	4871	124525	0.155	781722	3.39	3285	83971	4033	103093

Table A.3.38. Mercury content of yellow perch from OC83-4, a sediment-bottomed control limnocorral, 1983.

Date(s) sampled	Fish no.	Whole fish		Muscle tissue					Carcass			Combined muscle/carcass	
		Total length (mm)	Round wt (g)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	Total Hg (μg/g)	Specific activity (dpm/μg)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)
July 20,21	401	78	4.60	1.68	206	5266	0.150	35107	2.34	479	12244	365	9330
	402	79	4.72	1.65	198	5061	0.240	21088	2.48	488	12474	372	9509
	403	71	3.39	1.31	302	7720	0.140	55143	1.58	652	16667	493	12602
	404	68	3.15	1.15	387	9893	0.170	58194	1.64	699	17868	570	14571
	405	74	4.39	1.62	350	8947	0.640	13980	2.13	615	15721	501	12807
	406	90	6.83	2.49	166	4243	0.210	20205	3.58	434	11094	324	8282
Aug 10	407	75	4.80	1.84	1529	39085	0.970	40294	2.38	1315	33615	1408	35992
	408	84	6.33	2.24	2053	52480	0.940	55830	3.15	1535	39238	1750	44734
	409	84	5.79	2.29	2670	68252	1.380	49458	2.54	1900	48569	2265	57899
	410	86	6.71	2.62	1744	44581	1.130	39452	2.87	1270	32464	1496	38241
	411	83	6.10	2.49	1715	43839	0.800	54799	2.58	1547	39545	1630	41667
	412	86	6.72	2.69	1687	43124	1.090	39563	2.97	1483	37909	1580	40389
Aug 29	413	84	5.59	2.04	4261	108921	1.690	64450	2.48	2839	72572	3481	88983
	414	84	7.29	2.77	4206	107515	1.840	58432	3.12	2723	69606	3420	87423
	415	88	6.66	2.61	1947	49770	0.880	56557	2.93	1375	35148	1644	42025
	416	102	11.80	4.41	1382	35327	0.810	43614	4.83	1062	27147	1215	31058
	417	89	7.94	2.97	1735	44351	0.800	55439	3.01	1261	32234	1496	38241
	418	93	9.52	3.34	2380	60838	1.150	52903	4.01	1563	39954	1934	49438
	419	87	7.38	2.76	3115	79627	1.480	53802	2.91	2214	56595	2653	67817
	420	102	11.44	4.63	1415	36171	0.760	47593	5.10	1015	25946	1205	30803

Table A.3.39. Mean mercury content of yellow perch from OC83-4, a sediment-bottomed control limnocorral, 1983. (For sample sizes, see Table A.3.38).

Date(s) sampled	Fish no.	Whole fish		Muscle tissue					Carcass			Combined muscle/carcass	
		Total length (mm)	Round wt (g)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	Total Hg (μg/g)	Specific activity (dpm/μg)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)
June 30-July 10	Time 0	73	3.60	1.43			0.096						
July 20,21	401-406	77	4.51	1.65	268	6855	0.258	33953	2.29	561	14345	438	11184
Aug 10	407-412	83	6.08	2.36	1900	48560	1.052	46566	2.75	1508	38557	1688	43154
Aug 29	413-420	91	8.45	3.19	2555	65315	1.176	54099	3.55	1757	44900	2131	54474

Table A.3.40. Mercury content of yellow perch from OAS83-5, a sediment-bottomed limnocorral which had bank clay added in suspension in 1982, used without further treatment for the second consecutive year, 1983.

Date(s) sampled	Fish no.	Whole fish		Muscle tissue					Carcass			Combined muscle/carcass	
		Total length (mm)	Round wt (g)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	Total Hg (µg/g)	Specific activity (dpm/µg)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)
July 20,21	501	73	3.47	1.18	429	10966	0.080	137075	1.56	987	25230	747	19095
	502	82	5.39	1.73	215	5496	0.093	59097	2.95	617	15772	468	11963
	503	73	3.25	1.03	454	11605	0.057	203596	1.67	1061	27122	829	21191
	504	72	3.93	1.39	281	7183	0.110	65300	2.04	815	20833	599	15312
	505	76	4.22	1.57	229	5854	0.081	72272	1.97	732	18712	509	13011
	506	66	2.66	0.86	752	19223	0.071	270746	1.29	1387	35455	1133	28962
Aug 10	507	85	6.38	2.46	1240	31697	0.130	243823	2.92	1242	31748	1241	31723
	508	80	5.76	2.20	831	21242	0.110	193109	2.63	848	21677	840	21472
	509	79	5.11	2.05	1125	28758	0.100	287580	2.11	1195	30547	1161	29678
	510	95	9.62	3.78	468	11963	0.072	166153	4.26	613	15670	545	13931
	511	76	4.90	1.85	2033	51968	0.120	433067	2.17	1708	42661	1858	47495
	512	74	4.42	1.65	1003	25639	0.110	233082	1.97	992	25358	997	25486
Aug 29	513	89	7.03	2.66	3217	82234	0.140	587386	3.03	2472	63190	2820	72086
	514	105	12.70	5.00	3032	77505	0.180	430583	5.31	2115	54064	2560	65440
	515	86	6.19	2.51	2348	60020	0.110	545636	2.79	1849	47265	2085	53298
	516	96	8.57	3.12	2089	53400	0.130	410769	4.06	1462	37372	1734	44325
	517	85	6.25	2.40	3312	84663	0.140	604736	2.81	2204	56339	2714	69376
	518	89	6.95	2.54	3034	77556	0.150	517040	3.06	2124	54294	2537	64852
	519	86	7.07	2.84	2180	55726	0.110	506600	3.26	1759	44964	1955	49974
	520	97	9.42	3.66	3364	85992	0.170	505835	3.80	2326	59458	2835	72469
	521	93	8.56	3.32	1734	44325	0.094	471543	4.00	1357	34688	1528	39059
	522	102	11.54	4.33	965	24668	0.110	224255	4.89	911	23287	936	23926
	523	90	7.39	2.89	1118	28579	0.088	324761	3.19	1130	28885	1124	28732
	524	90	7.80	3.06	2124	54294	0.120	452450	3.47	1503	38420	1794	45859
	525	91	7.20	2.88	2855	72981	0.120	608175	3.28	2326	59458	2573	65772
	526	93	7.36	2.87	1501	38369	0.097	395557	3.54	1268	32413	1372	35072
	527	80	5.25	1.94	5460	139571	0.190	734584	2.27	3888	99387	4612	117894
	528	80	5.38	1.96	2810	71830	0.150	478867	2.39	2032	51943	2383	60915
	529	80	5.57	2.13	2461	62909	0.160	393181	2.75	1962	50153	2180	55726
	530	95	8.78	3.26	2301	58819	0.140	420136	4.03	1610	41155	1919	49054
	531	79	5.62	2.34	3624	92638	0.170	544929	2.40	2795	71447	3204	81902
	532	97	10.29	3.90	3089	78962	0.150	526413	4.56	2104	53783	2558	65389

Table A.3.41. Mean mercury content of yellow perch from OAS83-5, a sediment-bottomed limnocorral which had bank clay added in suspension in 1982, used without further treatment for the second consecutive year, 1983. (For sample sizes, see Table A.3.40).

Date(s) sampled	Fish no.	Whole fish		Muscle tissue					Carcass			Combined muscle/carcass	
		Total length (mm)	Round wt (g)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	Total Hg (μg/g)	Specific activity (dpm/μg)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)
June 30-July 10	Time 0	73	3.60	1.43			0.096						
July 20,21	501-506	74	3.82	1.29	393	10055	0.082	134681	1.91	933	23854	714	18256
Aug 10	507-512	82	6.03	2.33	1117	28545	0.107	259469	2.68	1100	28110	1107	28298
Aug 29	513-532	90	7.75	2.98	2631	67252	0.136	484172	3.44	1960	50098	2271	58056

Table A.3.42. Mercury content of yellow perch from OM83-6, a sediment-bottomed limnocorral with moss-peat added in 1981, used for the third consecutive year, 1983. Most of the fish were believed to have escaped through holes which developed in the wall of the limnocorral.

		Whole fish			Muscle tissue					Carcass		Combined muscle/carcass	
Date(s) sampled	Fish no.	Total length (mm)	Round wt (g)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	Total Hg (μg/g)	Specific activity (dpm/μg)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)
July 20,21	601	78	4.73	1.83	273	6979	0.099	70495	2.26	456	11656	374	9560
	602	78	4.66	1.93	196	5010	0.100	50100	2.33	470	12014	346	8845
	603	68	3.12	1.26	258	6595	0.079	83481	1.52	470	12014	374	9560
	604	70	3.20	1.33	310	7924	0.052	152385	1.55	549	14034	439	11222
MEANS:													
June 30- July 10	Time 0	73	3.60	1.43			0.096						
July 20,21	601-604	74	3.93	1.59	259	6627	0.083	89115	1.92	486	12430	383	9797

Table A.3.43. Mercury content of yellow perch from LM83-7, a sediment-bottomed limnocorral with added (low) moss, 1983.

Date(s) sampled	Fish no.	Whole fish		Muscle tissue					Carcass			Combined muscle/carcass	
		Total length (mm)	Round wt (g)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	Total Hg (μg/g)	Specific activity (dpm/μg)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)
July 20,21	701	81	5.40	2.09	176	4499	0.071	63366	2.62	454	11605	331	8461
	702	79	4.95	1.86	113	2889	0.096	30094	2.54	467	11938	317	8103
	703	80	4.77	1.69	241	6161	0.100	61610	2.41	509	13011	399	10199
	704	71	3.16	1.17	204	5215	0.080	65188	1.48	602	15389	426	10890
	705	63	2.31	0.80	321	8206	0.083	98867	1.06	763	19504	573	14647
Aug 10	706	89	7.15	2.81	1265	32336	0.095	340379	3.38	1017	25997	1130	28885
	707	76	4.20	1.55	1758	44939	0.100	449390	1.93	1312	33538	1511	38625
	708	78	4.47	1.69	2997	76610	0.110	696455	2.03	1821	46549	2355	60199
	709	76	4.66	1.60	2895	74003	0.120	616692	2.35	1916	48978	2313	59126
	710	73	3.59	1.35	1541	39392	0.110	358109	1.64	1185	30291	1346	34407
	711	91	7.17	2.93	1012	25869	0.140	184779	3.40	1050	26840	1032	26380
Aug 29	712	82	5.16	2.05	6223	159075	0.190	837237	2.10	3789	96856	4991	127582
	713	86	5.98	2.50	4456	113906	0.099	1150566	2.61	2584	66053	3500	89468
	714	91	7.15	2.97	4164	106442	0.120	887017	3.05	2663	68073	3404	87014
	715	90	7.41	2.92	6213	158819	0.130	1221685	2.84	4004	102352	5124	130982
	716	87	5.89	2.54	4176	106748	0.096	1111958	2.48	2726	69683	3460	88446
	717	90	7.83	3.11	2831	72367	0.120	603058	3.17	2035	52019	2429	62091
	718	90	6.71	2.68	4877	124668	0.120	1038900	2.85	3327	85046	4078	104243
	719	91	7.36	2.81	5078	129806	0.140	927186	3.34	3254	83180	4087	104473
	720	88	6.02	2.39	3545	90619	0.110	823809	2.70	2207	56416	2835	72469
	721	85	6.10	2.34	2498	63855	0.120	532125	2.55	1602	40951	2031	51917
	722	87	7.08	2.87	2390	61094	0.086	710395	2.90	1877	47981	2132	54499
	723	83	6.21	2.44	6018	153834	0.130	1183338	2.44	3930	100460	4974	127147
	724	92	7.56	3.06	3497	89392	0.088	1015818	3.40	2516	64315	2981	76201
	725	88	6.61	2.64	12413	317306	0.240	1322108	2.80	7469	190925	9868	252249
	726	79	4.99	2.01	4244	108487	0.100	1084870	2.17	2568	65644	3373	86222
	727	87	6.78	2.57	3865	98799	0.100	987990	3.26	2627	67152	3172	81084
	728	72	2.53	0.51	5851	149565	0.260	575250	1.25	3029	77428	3847	98338
	729	82	5.09	1.89	6398	163548	0.140	1168200	2.25	4189	107081	5197	132848
	730	78	4.62	1.79	4180	106851	0.150	712340	1.97	2959	75639	3540	90491
	731	87	6.35	2.24	5187	132592	0.110	1205382	3.01	3649	93277	4305	110046

Table A.3.44. Mean mercury content of yellow perch from LM83-7, a sediment-bottomed limnocorral with added (low) moss, 1983. (For sample sizes, see Table A.3.43).

Date(s) sampled	Fish no.	Whole fish		Muscle tissue					Carcass			Combined muscle/carcass	
		Total length (mm)	Round wt (g)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	Total Hg (µg/g)	Specific activity (dpm/µg)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)
June 30-July 10	Time 0	73	3.60	1.43			0.096						
July 20,21	701-705	75	4.12	1.52	211	5394	0.086	63825	2.02	559	14289	409	10460
Aug 10	706-711	81	5.21	1.99	1911	48858	0.113	440967	2.46	1384	35366	1615	41270
Aug 29	712-731	86	6.17	2.42	4905	125389	0.132	954962	2.66	3150	80527	3966	101391

Table A.3.45. Mercury content of yellow perch from LMS83-8, a sediment-bottomed limnocorral with added (low) moss and lake bottom sediment, 1983.

Date(s) sampled	Fish no.	Whole fish		Muscle tissue					Carcass			Combined muscle/carcass	
		Total length (mm)	Round wt (g)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	Total Hg (µg/g)	Specific activity (dpm/µg)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)
July 20,21	801	69	3.30	1.39	400	10225	0.059	173305	1.51	684	17485	548	14008
	802	72	3.28	1.18	378	9663	0.073	132370	1.63	579	14801	495	12653
	803	71	3.51	1.48	336	8589	0.062	138532	1.55	594	15184	468	11963
	804	63	2.38	0.93	312	7975	0.062	128629	1.05	486	12423	404	10327
	805	83	5.48	2.40	268	6851	0.078	87833	2.53	498	12730	386	9867
	806	65	2.24	0.73	420	10736	0.071	151211	1.12	679	17357	577	14749
Aug 10	807	78	4.70	1.81	7814	199744	0.150	1331627	2.20	4991	127582	6265	160148
	808	85	5.85	2.33	6177	157899	0.130	1214608	2.79	4227	108052	5114	130726
	809	84	6.42	2.12	4663	119197	0.130	916900	2.91	3427	87602	3948	100920
	810	80	5.05	1.92	5087	130036	0.120	1083633	2.21	3378	86350	4172	106646
	811	75	4.64	1.83	4071	104064	0.110	946036	2.26	2928	74847	3439	87909
	812	78	4.70	1.77	2934	75000	0.096	781250	2.28	2113	54013	2472	63190
Aug 29	813	93	7.76	2.99	9298	237679	0.170	1398112	3.40	5076	129755	7052	180266
	814	91	7.76	2.92	11859	303144	0.190	1595495	3.46	6534	167025	8971	229320
	815	93	8.36	3.35	10121	258717	0.150	1724780	3.60	6216	158896	8098	207004
	816	88	7.77	2.82	3962	101278	0.084	1205690	3.34	2905	74259	3389	86631
	817	92	7.76	2.99	10044	256748	0.170	1510282	3.39	6080	155419	7938	202914
	818	93	8.21	3.02	9510	243098	0.150	1620653	3.38	5899	150792	7603	194351
	819	96	8.99	3.25	11995	306621	0.180	1703450	3.96	7074	180828	9292	237526
	820	97	8.62	3.32	9719	248441	0.160	1552756	3.78	5641	144197	7548	192945
	821	84	5.82	2.23	6986	178579	0.120	1488158	2.57	4116	105215	5449	139289
	822	87	7.03	2.60	7947	203144	0.130	1562646	3.02	4690	119888	6197	158410
	823	81	5.79	2.04	13977	357285	0.200	1786425	2.64	8991	229831	11164	285378
	824	95	9.50	3.50	7912	202249	0.140	1444636	3.94	4977	127224	6358	162526
	825	80	5.65	2.03	9292	237526	0.140	1696614	2.32	6074	155266	7576	193661
	826	84	6.05	2.37	8243	210711	0.160	1316944	2.58	5662	144734	6898	176329
	827	80	4.99	1.83	5735	146600	0.140	1047143	2.03	3573	91334	4598	117536
	828	93	8.53	3.04	7212	184356	0.160	1152225	3.71	4995	127684	5993	153195
	829	79	4.79	1.76	6318	161503	0.120	1345858	2.24	4691	119913	5407	138216
	830	81	5.47	2.13	9603	245475	0.240	1022813	2.40	6011	153655	7700	196830
	831	83	6.15	2.32	11104	283845	0.210	1351643	2.67	5868	150000	8302	212219
	832	85	6.00	2.12	11940	305215	0.220	1387341	2.76	7455	190567	9403	240363

Table A.3.46. Mean mercury content of yellow perch from LMS83-8, a sediment-bottomed limnocorral with added (low) moss and lake bottom sediment, 1983. (For sample sizes, see Table A.3.45).

Date(s) sampled	Fish no.	Whole fish		Muscle tissue					Carcass			Combined muscle/carcass	
		Total length (mm)	Round wt (g)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	Total Hg (μg/g)	Specific activity (dpm/μg)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)
June 30-July 10	Time 0	73	3.60	1.43			0.096						
July 20,21	801-806	71	3.37	1.35	352	9007	0.068	135313	1.57	587	14997	480	12261
Aug 10	807-812	80	5.23	1.96	5124	130990	0.123	1045676	2.44	3511	89741	4235	108257
Aug 29	813-832	88	7.05	2.63	9139	233611	0.162	1445683	3.06	5626	143824	7247	185245

Table A.3.47. "Time zero" data on yellow perch, 1984.

		Whole fish		Muscle tissue		Carcass			Whole fish		Muscle tissue		Carcass
Date(s) sampled	Fish no.	Total length (mm)	Round wt (g)	Wt (g)	Total Hg (µg/g)	Wt (g)	Date(s) sampled	Fish no.	Total length (mm)	Round wt (g)	Wt (g)	Total Hg (µg/g)	Wt (g)
June 25	1	61	1.74	0.62	0.103	0.87	June 25	41	57	1.53			
	2	60	1.81	0.71	0.047	0.85		42	59	1.58			
	3	63	2.26	0.93	0.086	1.01		43	61	1.80			
	4	55	1.40	0.54	0.049	0.64		44	61	2.02			
	5	61	2.10	0.88	0.055	0.89		45	65	2.35			
	6	57	1.60	0.58	0.065	0.73		46	59	1.74			
	7	56	1.37	0.48	0.075	0.64		47	62	2.02			
	8	58	1.92	0.76	0.094	0.81		48	55	1.27			
	9	59	1.49	0.52	0.107	0.72		49	58	1.72			
	10	60	1.82	0.76	0.055	0.76		50	58	1.59			
	11	61	2.08	0.82	0.049	0.94		51	55	1.20			
	12	58	1.70	0.63	0.056	0.81		52	61	1.80			
	13	54	1.46	0.52	0.073	0.73		53	57	1.46			
	14	60	1.81	0.71	0.053	0.83		54	62	1.87			
	15	58	1.46	0.48	0.066	0.70		55	60	1.78			
	16	57	1.56	0.55	0.077	0.74		56	64	2.38			
	17	60	1.98	0.66	0.047	0.98		57	61	1.95			
	18	60	2.05	0.79	0.111	0.89		58	59	1.63			
	19	60	1.85	0.67	0.059	0.87		59	59	1.77			
	20	59	1.80	0.59	0.069	0.89		60	60	1.63			
	21	54	1.15	0.33	0.095	0.62		61	54	1.31			
	22	57	1.60	0.57	0.068	0.67		62	58	1.46			
	23	59	1.71	0.65	0.093	0.84		63	57	1.53			
	24	67	3.03	1.15	0.093	1.35		64	47	0.82			
	25	60	1.68					65	59	1.56			
	26	64	1.87					66	59	1.79			
	27	57	1.60					67	61	2.07			
	28	59	1.87					68	57	1.59			
	29	58	1.74					69	55	1.36			
	30	58	1.43					70	56	1.67			
	31	60	1.90					71	58	1.53			
	32	61	1.86					72	60	1.79			
	33	58	1.52					73	61	1.99			
	34	60	1.80					74	52	1.16			
	35	55	1.33					75	61	1.82			
	36	57	1.54					76	63	2.17			
	37	64	2.19					77	60	1.88			
	38	62	1.74					78	59	1.86			
	39	58	1.73										
	40	57	1.48										

Table A.3.48. Mercury content of yellow perch from LMN84-1, a sediment-bottomed limnocorral with added (low) moss, 1984. No radioisotope addition was made to this limnocorral.

		Whole fish		Muscle tissue					Carcass			Combined muscle/carcass	
Date(s) sampled	Fish no.	Total length (mm)	Round wt (g)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	Total Hg (µg/g)	Specific activity (dpm/µg)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)
July 12	101	64	2.12	0.78			0.057		0.96				
	102	60	1.93	0.69			0.067		1.16				
	103	63	2.10	0.79			0.073		1.07				
	104	58	1.80	0.60			0.091		0.89				
	105	62	2.23	0.74			0.058		1.11				
	106	61	1.92	0.60			0.057		1.05				
July 26	107	64	2.15	0.73			0.102		1.06				
	108	63	2.15	0.79			0.097		0.99				
	109	62	2.15	0.77			0.095		0.98				
	110	64	2.29	0.88			0.093		1.03				
	111	63	2.07	0.71			0.082		0.92				
	112	66	2.67	0.97			0.092		1.27				
	113	63	2.37	0.80			0.087		1.15				
	114	61	1.81	0.57			0.078		0.95				
	115	67	2.61	0.89			0.111		1.28				
	116	69	3.15	1.08			0.094		1.58				
Aug 2	117	67	2.79	1.05			0.079		1.43				
	118	68	3.05	1.12			0.067		1.51				
	119	72	3.56	1.31			0.111		1.82				
	120	74	3.86	1.41			0.093		1.89				
	121	61	2.04	0.81			0.086		0.92				
	122	69	2.73	1.10			0.096		1.34				
	123	67	2.87	1.14			0.109		1.35				
	124	59	1.87	0.72			0.089		0.95				
	125	62	2.21	0.79			0.100		1.09				
	126	63	2.01	0.81			0.113		0.94				
Aug 9	127	68	2.74	1.00			0.088		1.31				
	128	70	3.03	1.15			0.103		1.42				
	129	68	2.63	1.00			0.106		1.26				
	130	67	2.42	0.90			0.068		1.18				
	131	71	2.84	1.11			0.081		1.38				
	132	67	2.66	1.09			0.093		1.25				
	133	75	3.69	1.41			0.077		1.89				
	134	68	2.75	1.13			0.108		1.27				
	135	65	2.39	0.93			0.097		1.12				
	136	65	2.50	0.90			0.093		1.22				
Aug 16	137	71	3.33	1.31			0.082		1.56				
	138	65	2.34	0.91			0.131		1.06				
	139	64	2.29	0.87			0.095		1.06				
	140	75	3.77	1.43			0.103		1.89				
	141	71	3.07	1.23			0.121		1.51				
	142	68	2.67	1.03			0.117		1.23				
	143	82	4.45	1.61			0.076		2.27				
	144	66	2.47	0.97			0.095		1.21				
	145	67	2.60	1.02			0.133		1.20				
	146	68	2.76	1.05			0.145		1.24				
Aug 23	147	73	4.02	1.46			0.102		2.00				
	148	68	2.60	0.97			0.079		1.22				
	149	69	2.75	1.05			0.089		1.35				
	150	69	2.82	0.97			0.111		1.51				
	151	73	3.54	1.36			0.072		1.82				
	152	70	3.00	1.05			0.083		1.47				
	153	69	2.86	1.04			0.118		1.52				
	154	71	3.20	1.14			0.100		1.68				
	155	72	3.42	1.29			0.082		1.61				
	156	74	3.65	1.35			0.112		1.86				
Aug 29	157	62	2.28	0.87			0.141		1.07				
	158	70	3.25	1.25			0.104		1.52				
	159	69	2.82	1.03			0.117		1.48				
	160	72	3.07	1.11			0.114		1.62				
	161	67	2.68	0.99			0.142		1.31				
	162	76	4.07	1.56			0.122		2.02				
	163	73	3.55	1.31			0.141		1.65				
	164	72	3.14	1.09			0.160		1.68				
	165	69	3.00	1.14			0.168		1.43				
	166	69	2.71	1.01			0.148		1.37				
Sept 30	167	68	3.00	1.07			0.148		1.37				
	168	71	3.34	1.27			0.116		1.50				
	169	76	4.11	1.48			0.095		1.80				
	170	69	3.18	1.11			0.141		1.49				
	171	76	4.04	1.45			0.090		1.94				
	172	72	3.61	1.33			0.128		1.59				
	173	73	3.44	1.19			0.118		1.59				
	174	70	3.29	1.21			0.122		1.62				
	175	73	3.36	1.17			0.099		1.68				
	176	71	3.41	1.21			0.144		1.60				

Table A.3.49. Mean mercury content of yellow perch from LMN84-1, a sediment-bottomed limnocorral with added (low) moss, 1984. No radioisotope addition was made to this limnocorral. (For sample sizes, see Table A.3.48).

		Whole fish		Muscle tissue					Carcass			Combined muscle/carcass	
Date(s) sampled	Fish no.	Total length (mm)	Round wt (g)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	Total Hg (µg/g)	Specific activity (dpm/µg)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)
June 25	Time 0	59	1.72	0.66			0.073						
July 12	101-106	61	2.02	0.70			0.067		1.04				
July 26	107-116	64	2.34	0.82			0.093		1.12				
Aug 2	117-126	66	2.70	1.03			0.094		1.32				
Aug 9	127-136	68	2.77	1.06			0.091		1.33				
Aug 16	137-146	70	2.98	1.14			0.110		1.42				
Aug 23	147-156	71	3.19	1.17			0.095		1.60				
Aug 29	157-166	70	3.06	1.14			0.136		1.52				
Sept 30	167-176	72	3.48	1.25			0.120		1.62				

Table A.3.50. Mercury content of yellow perch from CN84-2, a sediment-bottomed control limnocorral, 1984. No radioisotope addition was made to this limnocorral.

		Whole fish		Muscle tissue					Carcass			Combined muscle/carcass	
Date(s) sampled	Fish no.	Total length (mm)	Round wt (g)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	Total Hg (μg/g)	Specific activity (dpm/μg)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)
July 12	201	59	2.03	0.75			0.040		0.98				
	202	57	1.80	0.65			0.083		0.83				
	203	59	2.01	0.82			0.057		0.98				
	204	60	1.79	0.66			0.077		0.92				
	205	61	2.13	0.79			0.060		1.08				
	206	56	1.54	0.54			0.062		0.78				
July 26	207	65	2.41	0.79			0.085		1.11				
	208	64	2.46	0.84			0.073		1.19				
	209	71	3.51	1.36			0.065		1.60				
	210	59	1.82	0.61			0.072		0.88				
	211	62	1.85	0.65			0.054		0.84				
	212	58	1.57	0.55			0.056		0.68				
	213	61	1.97	0.58			0.061		0.99				
	214	59	1.50	0.49			0.061		0.75				
	215	59	1.60	0.59			0.073		0.81				
	216	66	2.46	0.94			0.063		1.20				
Aug 2	217	68	2.74	1.05			0.082		1.28				
	218	76	4.24	1.56			0.077		2.18				
	219	65	2.36	0.93			0.070		1.09				
	220	67	2.54	1.01			0.046		1.25				
	221	72	3.36	1.37			0.053		1.67				
	222	58	1.73	0.62			0.059		0.82				
	223	54	1.28	0.38			0.064		0.67				
	224	60	1.50	0.48			0.096		0.78				
	225	64	2.39	0.91			0.079		1.21				
	226	61	1.95	0.75			0.081		0.92				
Aug 9	227	67	2.98	1.04			0.080		1.41				
	228	65	2.27	0.80			0.046		1.05				
	229	64	2.07	0.72			0.064		1.02				
	230	58	1.70	0.59			0.068		0.81				
	231	71	3.13	1.19			0.063		1.44				
	232	65	2.39	0.85			0.053		1.22				
	233	67	2.75	1.04			0.054		1.32				
	234	69	2.59	0.95			0.074		1.22				
	235	58	1.62	0.50			0.057		0.84				
	236	58	1.87	0.77			0.041		0.90				
Aug 16	237	65	2.74	1.09			0.052		1.29				
	238	60	1.92	0.68			0.044		0.86				
	239	65	2.00	0.54			0.065		1.03				
	240	60	1.95	0.75			0.056		0.83				
	241	63	2.24	0.81			0.044		1.06				
	242	58	1.90	0.68			0.052		0.85				
	243	65	2.44	0.94			0.044		1.13				
	244	61	2.06	0.71			0.063		0.99				
	245	68	2.76	1.12			0.046		1.33				
	246	49	1.11	0.43			0.052		0.44				
Aug 23	247	75	3.77	1.57			0.047		1.72				
	248	61	1.91	0.63			0.049		1.03				
	249	67	2.47	0.90			0.055		1.30				
	250	73	3.53	1.31			0.050		1.60				
	251	68	2.57	0.94			0.061		1.36				
	252	63	2.23	0.79			0.048		1.09				
	253	53	1.31	0.53			0.047		0.55				
	254	68	2.74	0.99			0.068		1.37				
	255	67	2.41	0.89			0.059		1.21				
	256	54	1.48	0.58			0.061		0.65				
Aug 29	257	75	3.77	1.52			0.082		1.81				
Sept 30	258	60	1.98	0.75			0.055		0.91				
	259	60	1.99	0.69			0.063		0.95				
	260	70	3.08	1.18			0.057		1.31				
	261	59	1.88	0.71			0.050		0.82				
	262	62	2.13	0.74			0.052		0.95				
	263	61	2.12	0.78			0.086		1.07				
	264	59	1.96	0.71			0.077		0.85				
	265	59	1.90	0.67			0.088		0.85				
	266	79	4.78	1.91			0.066		2.22				
	267	88	6.65	2.60			0.093		2.99				

Table A.3.51. Mean mercury content of yellow perch from CN84-2, a sediment-bottomed control limnocorral, 1984. No radioisotope addition was made to this limnocorral. (For sample sizes, see Table A.3.50).

		Whole fish		Muscle tissue					Carcass			Combined muscle/carcass	
Date(s) sampled	Fish no.	Total length (mm)	Round wt (g)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	Total Hg (μg/g)	Specific activity (dpm/μg)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)
June 25	Time 0	59	1.72	0.66			0.073						
July 12	201-206	59	1.88	0.70			0.063		0.93				
July 26	207-216	62	2.12	0.74			0.066		1.01				
Aug 2	217-226	65	2.41	0.91			0.071		1.19				
Aug 9	227-236	64	2.34	0.85			0.060		1.12				
Aug 16	237-246	61	2.11	0.78			0.052		0.98				
Aug 23	247-256	65	2.44	0.91			0.055		1.19				
Aug 29	257	75	3.77	1.52			0.082		1.81				
Sept 30	258-267	66	2.85	1.07			0.069		1.29				

Table A.3.52. Mercury content of yellow perch from SP84-3, a sediment-bottomed limnocorral with added black spruce boughs, 1984.

		Whole fish		Muscle tissue					Carcass			Combined muscle/carcass	
Date(s) sampled	Fish no.	Total length (mm)	Round wt (g)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	Total Hg (ug/g)	Specific activity (dpm/ug)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)
July 12	301	68	2.99	1.15	232	5930	0.099	59899	1.49	540	13804	406	10378
	302	61	2.11	0.74	360	9202	0.088	104568	1.13	660	16871	541	13829
	303	55	1.74	0.63	398	10174	0.059	172441	0.90	805	20578	637	16283
	304	57	1.71	0.55	456	11656	0.065	179323	0.94	1147	29320	892	22802
	305	61	2.24	0.83	298	7618	0.088	86568	1.12	819	20936	597	15261
	306	58	1.76	0.49	504	12883	0.106	121538	0.89	1006	25716	828	21166
July 26	307	65	2.64	0.90	2532	64724	0.094	688553	1.40	2342	59867	2416	61759
	308	60	1.81	0.62	2171	55496	0.087	637885	0.96	2343	59893	2276	58180
	309	69	3.01	1.06	2277	58206	0.072	808417	1.53	2333	59637	2310	59049
	310	63	2.16	0.77	2816	71984	0.076	947158	1.01	2770	70808	2790	71319
	311	63	2.33	0.84	2422	61912	0.089	695640	1.16	2308	58998	2356	60225
	312	61	1.95	0.64	3153	80598	0.120	671650	0.96	2629	67203	2839	72572
	313	61	2.02	0.74	3731	95373	0.098	973194	0.97	3425	87551	3557	90925
	314	66	2.48	0.87	1845	47163	0.055	857509	1.30	1863	47623	1856	47444
	315	68	2.76	0.88	2265	57899	0.081	714802	1.46	2375	60711	2334	59663
	316	66	2.61	0.97	2666	68149	0.087	783322	1.23	2450	62628	2545	65056
Aug 2	317	70	2.91	1.13	2879	73594	0.101	728653	1.41	2203	56314	2504	64008
	318	61	1.91	0.73	5668	144888	0.107	1354093	0.88	3731	95373	4609	117817
	319	69	2.71	1.01	3392	86708	0.104	833731	1.33	2810	71830	3061	78246
	320	67	2.61	0.94	3023	77275	0.094	822074	1.27	1964	50204	2414	61708
	321	64	2.18	0.89	3545	90619	0.111	816387	1.05	2982	76227	3240	82822
	322	72	3.37	1.35	1963	50179	0.067	748940	1.63	1570	40133	1748	44683
	323	65	2.60	0.98	4975	127173	0.089	1428910	1.33	3369	86120	4050	103528
	324	64	2.15	0.85	3408	87117	0.069	1262565	1.03	2584	66053	2957	75588
	325	61	2.09	0.86	6011	153655	0.106	1449575	0.96	4136	105726	5022	128374
	326	65	2.50	0.96	5206	133078	0.111	1198901	1.24	3860	98671	4447	113676
Aug 9	327	75	3.44	1.14	4268	109100	0.102	1069608	1.78	2888	73824	3427	87602
	328	78	4.12	1.57	2199	56212	0.043	1307256	2.02	1704	43558	1920	49080
	329	71	3.11	1.12	3877	99105	0.073	1357603	1.59	2574	65798	3113	79576
	330	70	2.96	1.25	4966	126943	0.075	1692573	1.30	3345	85506	4140	105828
	331	61	2.00	0.73	7501	191743	0.093	2061753	0.87	4726	120808	5992	153170
	332	67	2.95	1.07	5012	128119	0.081	1581716	1.36	3447	88113	4136	105726
	333	65	2.45	0.97	5488	140286	0.082	1710805	1.02	3412	87219	4424	113088
	334	69	3.17	1.12	8304	212270	0.115	1845826	1.62	5683	145271	6754	172648
	335	66	2.66	1.06	4377	111887	0.108	1035991	1.20	2954	75511	3621	92561
	336	70	2.87	1.13	5690	145450	0.098	1484184	1.43	4005	102377	4749	121396
Aug 16	337	70	2.88	1.18	7698	196779	0.117	1681872	1.46	5813	148594	6656	170143
	338	64	2.39	0.91	7475	191079	0.108	1769250	1.03	5013	128144	6168	157669
	339	72	3.14	1.24	6371	162858	0.119	1368555	1.59	3859	98645	4960	126789
	340	73	3.82	1.47	11100	283742	0.134	2117478	1.84	6831	174617	8727	223083
	341	76	4.16	1.64	4314	110276	0.087	1267540	2.07	3007	76866	3585	91641
	342	70	3.10	1.26	6069	155138	0.088	1762932	1.50	3707	94760	4785	122316
	343	63	2.35	0.92	12854	328579	0.175	1877594	1.11	8031	205291	10217	261171
	344	71	3.39	1.37	6355	162449	0.088	1846011	1.50	4189	107081	5223	133512
	345	72	3.42	1.43	5092	130164	0.105	1239657	1.56	3284	83947	4149	106058
	346	67	2.75	1.16	8787	224617	0.134	1676246	1.31	4546	116207	6538	167127
Aug 23	347	72	3.14	1.29	12036	307669	0.145	2121855	1.55	8061	206058	9867	252224
	348	77	4.12	1.55	8675	221754	0.118	1879271	1.98	5734	146575	7025	179576
	349	72	3.53	1.40	6538	167127	0.102	1638500	1.67	3825	97776	5062	129397
	350	73	3.37	1.41	12047	307950	0.158	1949051	1.41	7667	195987	9857	251968
	351	73	3.50	1.40	7699	196805	0.105	1874333	1.69	5016	128221	6232	159305
	352	74	3.55	1.38	9780	250000	0.127	1968504	1.71	6141	156979	7766	198517
	353	72	3.09	1.18	11120	284254	0.136	2090103	1.51	7282	186145	8966	229192
	354	77	3.90	1.58	7292	186401	0.104	1792317	1.90	4946	126431	6011	153655
	355	70	2.94	1.19	9170	234407	0.116	2020750	1.38	5880	150307	7403	189238
	356	73	3.88	1.49	8638	220808	0.119	1855529	1.81	5582	142689	6962	177965
Aug 29	357	69	3.37	1.27	6932	177198	0.118	1501678	1.54	4132	105624	5397	137960
	358	77	3.83	1.36	16557	423236	0.189	2239344	1.81	10313	263625	12992	332106
	359	70	3.05	1.05	10618	271421	0.138	1966819	1.61	5950	152096	7793	199208
	360	74	3.45	1.28	15332	391922	0.180	2177344	1.68	10111	258461	12369	316181
	361	75	3.75	1.39	17410	445041	0.206	2160393	1.81	10279	262756	13377	341948
	362	66	2.69	1.06	13043	333410	0.172	1938430	1.21	7438	190133	10055	257030
	363	76	3.76	1.49	9597	245322	0.119	2061529	1.70	6033	154218	7698	196779
	364	79	4.31	1.55	8364	213804	0.127	1683496	2.15	5513	140925	6707	171447
	365	74	3.86	1.46	17767	454167	0.243	1869000	1.89	11252	287628	14091	360199
	366	80	4.27	1.65	13393	342357	0.173	1978942	2.00	8506	217434	10715	273901
Sept 30	367	78	4.72	1.73	8712	222699	0.109	2043110	2.25	4558	116513	6364	162679
	368	77	4.47	1.56	25792	659305	0.256	2575410	2.00	13564	346728	18921	483666
	369	79	4.48	1.55	17566	449029	0.200	2245145	2.29	9432	241104	12715	325026
	370	78	4.75	1.77	13861	354320	0.169	2096568	2.21	8394	214571	10825	276713
	371	80	5.07	1.77	11029	281927	0.142	1985401	2.55	6486	165798	8347	213369
	372	80	4.77	1.70	18302	467843	0.219	2136269	2.24	10296	263190	13750	351483
	373	80	4.64	1.82	15213	388880	0.179	2172514	2.14	8408	214928	11536	294888
	374	79	4.63	1.65	20341	519964	0.232	2241224	2.15	10926	279294	15014	383793
	375	81	5.10	1.84	15110	386247	0.188	2054505	2.45	8480	216769	11324	289468
	376	85	5.31	2.08	17854	456391	0.227	2010533	2.58	9727	248645	13355	341385

Table A.3.53. Mean mercury content of yellow perch from SP84-3, a sediment-bottomed limnocorral with added black spruce boughs, 1984. (For sample sizes, see Table A.3.52).

Date(s) sampled	Fish no.	Whole fish		Muscle tissue					Carcass			muscle/carcass	
		Total length (mm)	Round weight (g)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	Total Hg (µg/g)	Specific activity (dpm/µg)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)
June 25	Time 0	59	1.72	0.66			0.073						
July 12	301-306	60	2.09	0.73	375	9577	0.084	120723	1.08	830	21204	650	16620
July 26	307-316	64	2.38	0.83	2588	66150	0.086	777813	1.20	2484	63492	2528	64619
Aug 2	317-326	66	2.50	0.97	4007	102429	0.096	1064383	1.21	2921	74665	3405	87045
Aug 9	327-336	69	2.97	1.12	5168	132112	0.087	1514732	1.42	3474	88799	4228	108068
Aug 16	337-346	70	3.14	1.26	7612	194568	0.116	1660714	1.50	4828	123415	6101	155951
Aug 23	347-356	73	3.50	1.39	9300	237718	0.123	1919021	1.66	6013	153717	7515	192104
Aug 29	357-366	74	3.63	1.36	12901	329788	0.167	1957698	1.74	7953	203290	10119	258676
Sept 30	367-376	80	4.79	1.75	16378	418661	0.192	2156068	2.29	9027	230754	12215	312247

Table A.3.54. Mercury content of yellow perch from SD84-4, a sediment-bottomed limnocorral with added prairie sod, 1984.

		Whole fish		Muscle tissue					Carcass			Combined muscle/carcass	
Date(s) sampled	Fish no.	Total length (mm)	Round wt (g)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	Total Hg (µg/g)	Specific activity (dpm/µg)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)
July 12	401	61	2.15	0.82	416	10634	0.054	196926	1.11	783	20015	627	16028
	402	63	2.26	0.84	369	9433	0.059	159881	1.14	735	18788	580	14826
	403	64	2.61	0.98	465	11887	0.045	264156	1.36	918	23466	728	18609
	404	58	1.78	0.61	399	10199	0.061	167197	0.83	713	18226	580	14826
	405	61	2.04	0.68	519	13267	0.073	181740	1.12	871	22265	738	18865
	406	66	2.80	1.05	301	7694	0.060	128233	1.40	721	18430	541	13829
July 26	407	64	2.54	0.91	1684	43047	0.054	797167	1.11	1534	39213	1602	40951
	408	67	2.79	0.97	1751	44760	0.060	746000	1.35	1576	40286	1649	42152
	409	75	4.62	1.77	1635	41794	0.058	720586	2.27	1434	36656	1522	38906
	410	69	3.23	1.07	1661	42459	0.063	673952	1.57	1647	42101	1653	42255
	411	62	2.33	0.89	2547	65107	0.058	1122534	1.16	2030	51892	2254	57618
	412	72	3.74	1.43	1531	39136	0.057	686596	1.68	1545	39494	1539	39340
	413	71	3.52	1.42	1471	37602	0.042	895286	1.36	1245	31825	1360	34765
	414	73	4.12	1.61	1662	42485	0.049	867041	1.74	1433	36631	1543	39443
	415	72	3.53	1.31	1619	41385	0.052	795865	1.68	1441	36835	1519	38829
	416	64	2.29	0.81	1522	38906	0.061	637803	1.01	1267	32388	1380	35276
Aug 2	417	75	4.19	1.70	3034	77556	0.050	1551120	2.01	2250	57515	2609	66692
	418	83	5.13	2.14	2220	56748	0.038	1493368	2.31	1629	41641	1913	48901
	419	74	4.37	1.74	1957	50026	0.041	1220146	2.01	1657	42357	1796	45910
	420	71	3.40	1.35	2221	56774	0.052	1091808	1.54	1819	46498	2007	51304
	421	78	4.76	1.95	1908	48773	0.042	1161262	2.11	1539	39340	1716	43865
	422	80	4.96	2.03	2495	63778	0.056	1138893	2.33	1757	44913	2101	53707
	423	77	4.76	2.12	2034	51994	0.039	1333179	2.01	1501	38369	1775	45373
	424	72	3.97	1.58	3160	80777	0.049	1648510	1.72	2290	58538	2707	69197
	425	82	5.34	2.19	3030	77454	0.061	1269738	2.45	2070	52914	2523	64494
	426	83	5.63	2.47	2242	57311	0.040	1432775	2.39	1679	42919	1965	50230
Aug 9	427	86	6.17	2.58	5463	139647	0.093	1501581	2.99	3917	100128	4633	118430
	428	88	7.65	2.94	4293	109739	0.065	1688292	3.55	3240	82822	3717	95015
	429	88	6.86	2.97	4377	111887	0.079	1416291	3.06	3244	82924	3802	97188
	430	82	6.33	2.58	3453	88267	0.056	1576196	2.65	2817	72009	3131	80036
	431	74	3.89	1.51	3255	83206	0.059	1410271	1.74	2328	59509	2759	70527
	432	79	5.28	2.04	4426	113139	0.062	1824823	2.42	3105	79371	3709	94811
	433	80	4.97	2.05	4555	116437	0.064	1819328	2.16	2992	76483	3753	95936
	434	84	5.96	2.56	3449	88165	0.061	1445328	2.51	2573	65772	3015	77071
	435	82	6.22	2.43	4149	106058	0.070	1515114	2.71	2978	76125	3532	90286
	436	77	5.28	2.22	4032	103067	0.062	1662371	2.15	3129	79985	3588	91718
Aug 16	437	80	5.19	2.24	6170	157720	0.083	1900241	2.16	4036	103170	5122	130930
	438	85	6.05	2.74	5621	143686	0.066	2177061	2.38	3720	95092	4737	121089
	439	83	5.65	2.36	5172	132209	0.069	1916072	2.64	3075	78604	4065	103911
	440	85	7.34	2.75	5443	139136	0.081	1717728	3.55	3695	94453	4458	113957
	441	85	6.25	2.60	4664	119223	0.079	1509152	2.99	3093	79064	3824	97751
	442	79	4.73	2.01	5605	143277	0.089	1609854	2.05	3695	94453	4641	118635
	443	84	5.55	2.47	5090	130112	0.084	1548952	2.48	3044	77812	4065	103911
	444	91	7.23	3.13	4320	110429	0.081	1363321	3.52	2873	73441	3554	90849
	445	85	6.25	2.43	5181	132439	0.079	1676443	2.92	3101	79269	4045	103400
	446	89	7.33	3.18	5331	136273	0.089	1531157	3.32	3656	93456	4475	114392
Aug 23	447	96	8.28	3.47	5050	129090	0.088	1466932	4.05	2971	75946	3930	100460
	448	86	6.01	2.44	7929	202684	0.101	2006772	2.80	4822	123262	6269	160251
	449	81	5.36	2.24	6029	154116	0.094	1639532	2.27	3609	92255	4811	122981
	450	93	7.41	3.06	5556	142025	0.091	1560714	3.21	3204	81902	4352	111247
	451	85	6.40	2.45	5779	147725	0.105	1406905	2.98	3732	95399	4656	119018
	452	84	5.50	2.18	9296	237628	0.137	1734511	2.68	5583	142715	7249	185302
	453	86	6.79	2.97	6128	156646	0.104	1506212	2.75	3706	94734	4964	126892
	454	87	6.29	2.62	6281	160557	0.116	1384112	2.95	3888	99387	5014	128170
	455	84	5.59	2.31	7086	181135	0.133	1361917	2.65	4428	113190	5666	144836
	456	93	7.61	3.22	6025	154013	0.106	1452953	3.35	3738	95552	4859	124208
Aug 29	457	89	6.79	2.68	6412	163906	0.108	1517648	3.06	3944	100818	5096	130266
	458	92	7.17	3.00	6228	159202	0.094	1693638	3.45	3543	90567	4792	122495
	459	94	8.85	3.30	4579	117050	0.081	1445062	4.16	2650	67740	3503	89545
Sept 30	460	94	8.18	3.26	5776	147648	0.092	1604870	3.71	3484	89059	4556	116462
	461	91	7.57	2.82	7011	179218	0.093	1927075	3.13	4133	105649	5497	140516
	462	94	8.27	3.28	5699	145680	0.076	1916842	3.93	3258	83282	4368	111656
	463	92	7.59	2.99	6498	166104	0.099	1677818	3.28	3486	89110	4922	125818
	464	88	6.94	2.70	7830	200153	0.099	2021747	3.16	4104	104908	5821	148799
	465	93	8.18	3.19	6861	175383	0.098	1789622	3.36	3896	99591	5340	136503
	466	84	6.45	2.61	9296	237628	0.126	1885937	2.56	4918	125716	7128	182209
	467	87	7.26	2.97	6590	168456	0.119	1415597	3.37	3721	95118	5065	129473
	468	79	5.10	1.95	5572	142434	0.094	1515255	2.19	3404	87014	4425	113113
	469	84	5.52	2.23	7524	192331	0.104	1849337	2.55	3998	102198	5643	144248

Table A.3.55. Mean mercury content of yellow perch from SD84-4, a sediment-bottomed limnocorral with added prairie sod, 1984. (For sample sizes, see Table A.3.54).

		Whole fish		Muscle tissue					Carcass			Combined muscle/carcass	
Date(s) sampled	Fish no.	Total length (mm)	Round wt (g)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	Total Hg (μg/g)	Specific activity (dpm/μg)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)
June 25	Time 0	59	1.72	0.66			0.073						
July 12	401-406	62	2.27	0.83	412	10519	0.059	183022	1.16	790	20198	632	16164
July 26	407-416	69	3.27	1.22	1708	43668	0.055	794283	1.49	1515	38732	1602	40954
Aug 2	417-426	78	4.65	1.93	2430	62119	0.047	1334080	2.09	1819	46500	2111	53967
Aug 9	427-436	82	5.86	2.39	4145	105961	0.067	1585960	2.59	3032	77513	3564	91102
Aug 16	437-446	85	6.16	2.59	5260	134450	0.080	1694998	2.80	3399	86881	4299	109883
Aug 23	447-456	88	6.52	2.70	6516	166562	0.108	1552056	2.97	3968	101434	5177	132337
Aug 29	457-459	92	7.60	2.99	5740	146719	0.094	1552116	3.56	3379	86375	4464	114102
Sept 30	460-469	89	7.11	2.80	6866	175504	0.100	1760410	3.12	3840	98165	5277	134880

Table A.3.56. Mercury content of yellow perch from LM84-5, a sediment-bottomed limnocorral with added (low) moss, 1984.

		Whole fish		Muscle tissue					Carcass			Combined muscle/carcass	
Date(s) sampled	Fish no.	Total length (mm)	Round wt (g)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	Total Hg (µg/g)	Specific activity (dpm/µg)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)
July 12	501	62	2.04	0.69	257	6570	0.095	69158	1.04	553	14136	435	11120
	502	67	2.85	1.05	211	5394	0.064	84281	1.49	448	11452	350	8947
	503	60	1.68	0.60	445	11375	0.053	214623	0.90	656	16769	572	14622
	504	60	1.80	0.60	419	10711	0.055	194745	0.94	790	20194	645	16488
	505	54	1.27	0.38	453	11580	0.062	186774	0.69	998	25511	804	20552
	506	61	1.67	0.52	405	10353	0.068	152250	0.84	702	17945	588	15031
July 26	507	68	2.80	0.97	2184	55828	0.107	521757	1.34	1741	44504	1927	49259
	508	61	2.06	0.71	1742	44530	0.080	556625	0.93	1670	42689	1701	43482
	509	62	2.06	0.64	2023	51713	0.116	445802	0.95	1646	42076	1798	45961
	510	61	2.04	0.67	2495	63778	0.106	601679	0.98	1924	49182	2156	55112
	511	62	2.13	0.81	2167	55394	0.083	667398	0.91	1846	47188	1997	51048
	512	68	2.64	0.92	1631	41692	0.100	416920	1.27	1225	31314	1396	35685
	513	63	2.37	0.84	1875	47929	0.110	435718	1.05	1544	39468	1691	43226
	514	65	2.29	0.76	1356	34663	0.087	398425	1.15	1063	27173	1180	30164
	515	62	2.12	0.77	2073	52991	0.103	514476	0.96	1804	46115	1924	49182
Aug 2	516	67	2.57	1.00	2500	63906	0.089	718045	1.16	1857	47469	2155	55087
	517	66	2.63	1.01	2218	56697	0.082	691427	1.23	1735	44351	1953	49923
	518	65	2.48	0.99	2473	63216	0.088	718364	1.18	1781	45527	2097	53604
	519	64	2.30	0.88	2286	58436	0.082	712634	1.10	1732	44274	1978	50562
	520	68	2.92	1.11	2527	64596	0.081	797481	1.46	1988	50818	2221	56774
	521	69	2.71	0.97	1426	36452	0.095	383705	1.39	1171	29934	1276	32618
	522	67	2.52	0.96	2699	68993	0.094	733968	1.18	1988	50818	2307	58972
	523	62	2.16	0.82	3084	78834	0.110	716673	0.96	2165	55343	2588	66155
	524	64	2.46	0.97	2904	74233	0.082	905280	0.99	2075	53042	2485	63522
	525	65	2.51	0.93	2723	69606	0.113	615982	1.17	1736	44376	2173	55547
Aug 9	526	68	2.72	1.08	3351	85660	0.091	941319	1.25	2101	53707	2680	68507
	527	71	3.21	1.27	2855	72981	0.078	935654	1.50	1602	40951	2176	55624
	528	66	2.54	1.05	2795	71447	0.095	752074	1.22	1971	50383	2352	60123
	529	72	3.19	1.15	2698	68967	0.101	682842	1.64	1711	43737	2118	54141
	530	63	2.32	0.93	2445	62500	0.088	710227	1.07	1605	41028	1996	51022
	531	60	2.09	0.83	2865	73236	0.079	927038	0.91	1849	47265	2334	59663
	532	72	3.39	1.29	2001	51150	0.080	639375	1.62	1394	35634	1663	42510
	533	67	2.81	1.12	3309	84586	0.097	872021	1.34	2037	52071	2616	66871
	534	63	2.25	0.89	3766	96268	0.103	934641	1.04	2204	56339	2924	74744
	535	65	2.41	0.87	3048	77914	0.096	811604	1.12	1842	47086	2369	60557
Aug 16	536	75	4.13	1.43	1803	46089	0.072	640125	2.09	1049	26815	1355	34637
	537	70	3.23	1.29	4581	117101	0.118	992381	1.45	2694	68865	3582	91564
	538	78	4.56	1.76	2999	76662	0.106	723226	2.25	1835	46907	2346	59969
	539	66	2.75	1.11	3194	81646	0.077	1060338	1.26	1961	50128	2538	64877
	540	68	3.11	1.26	3092	79039	0.088	898170	1.41	1994	50971	2512	64213
	541	70	3.55	1.36	4448	113701	0.124	916944	1.73	2641	67510	3436	87832
	542	76	3.89	1.56	2426	62014	0.073	849507	1.91	1798	45961	2080	53170
	543	70	3.32	1.28	4041	103298	0.091	1135143	1.63	2602	66513	3235	82694
	544	72	3.42	1.45	3463	88522	0.100	885220	1.64	2080	53170	2730	69785
	545	71	3.42	1.36	3490	89213	0.121	737298	1.66	2047	52326	2697	68942
Aug 23	546	74	3.49	1.42	3686	94223	0.125	753784	1.66	2425	61989	3006	76840
	547	67	2.78	1.05	4191	107132	0.115	931583	1.32	2602	66513	3306	84509
	548	73	3.78	1.44	4197	107285	0.124	865202	1.78	2241	57285	3116	79652
	549	76	4.09	1.48	3970	101483	0.130	780638	2.07	2402	61401	3056	78119
	550	72	3.34	1.35	4509	115261	0.114	1011061	1.54	2550	65184	3465	88574
	551	77	4.01	1.50	3768	96319	0.144	668882	1.93	2312	59100	2949	75383
	552	68	2.81	1.11	5452	139366	0.151	922954	1.34	3174	81135	4206	107515
	553	71	3.57	1.42	3933	100537	0.118	852008	1.69	2485	63522	3146	80419
	554	73	3.52	1.32	3696	94479	0.119	793941	1.69	2367	60506	2950	75409
	555	76	3.99	1.43	3712	94888	0.155	612181	1.87	2323	59381	2925	74770
Aug 29	556	77	4.37	1.51	4381	111989	0.115	973817	2.12	2409	61580	3229	82541
	557	74	3.47	1.37	5428	138753	0.135	1027800	1.72	2935	75026	4040	103272
	558	80	4.32	1.67	5025	128451	0.137	937599	2.03	2480	63395	3629	92766
	559	74	3.59	1.40	4846	123875	0.129	960271	1.76	2765	70680	3687	94248
	560	76	3.94	1.62	5184	132515	0.134	988918	1.82	2974	76022	4015	102633
	561	76	4.12	1.50	5752	147035	0.184	799103	2.07	3087	78911	4207	107541
	562	79	4.17	1.59	5407	138216	0.137	1008876	2.10	2820	72086	3935	100588
	563	72	3.15	1.17	5453	139392	0.133	1048060	1.60	2765	70680	3900	99693
	564	74	3.77	1.39	5431	138829	0.151	919397	1.74	3279	83819	4235	108257
Sept 30	565	77	4.17	1.54	5703	145782	0.149	978403	2.06	2922	74693	4112	105112
	566	73	3.75	1.45	6195	158359	0.165	959752	1.76	3110	79499	4504	115133
	567	76	4.26	1.50	6577	168124	0.187	899059	1.89	3592	91820	4913	125588
	568	75	4.58	1.67	5358	136963	0.153	895183	2.14	2942	75204	4001	102275

Table A.3.57. Mean mercury content of yellow perch from LM84-5, a sediment-bottomed limnocorral with added (low) moss, 1984. (For sample sizes, see Table A.3.56).

		Whole fish		Muscle tissue					Carcass			Combined muscle/carcass	
Date(s) Sampled	Fish no.	Total length (mm)	Round wt (g)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	Total Hg (μg/g)	Specific activity (dpm/μg)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)
June 25	Time 0	59	1.72	0.66			0.073						
July 12	501-506	61	1.89	0.64	365	9331	0.066	150305	0.98	691	17668	566	14460
July 26	507-515	64	2.28	0.79	1950	49835	0.099	506533	1.06	1607	41079	1752	44791
Aug 2	516-525	66	2.53	0.96	2484	63497	0.092	699356	1.18	1823	46595	2123	54276
Aug 9	526-535	67	2.69	1.05	2913	74471	0.091	820680	1.27	1832	46820	2323	59376
Aug 16	536-545	72	3.54	1.39	3354	85729	0.097	883835	1.70	2070	52917	2651	67768
Aug 23	546-555	73	3.54	1.35	4111	105097	0.130	819223	1.69	2488	63602	3213	82119
Aug 29	556-564	76	3.88	1.47	5212	133228	0.139	962649	1.88	2835	72467	3875	99060
Sept 30	565-568	75	4.19	1.54	5958	152307	0.164	933099	1.96	3142	80304	4383	112027

Table A.3.58. Mercury content of yellow perch from CHg84-6, a sediment-bottomed control limnocorral with both "cold" (ionic) mercury and "hot" ^{203}Hg, 1984.

		Whole fish		Muscle tissue					Carcass			Combined muscle/carcass	
Date(s) sampled	Fish no.	Total length (mm)	Round wt (g)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	Total Hg (μg/g)	Specific activity (dpm/μg)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)
July 12	601	59	1.73	0.60	376	9611	0.072	133486	0.90	1070	27352	792	20245
	602	61	2.07	0.73	269	6876	0.055	125018	1.02	838	21421	601	15363
	603	58	1.93	0.70	313	8001	0.088	90920	1.04	795	20322	601	15363
	604	62	1.84	0.70	343	8768	0.060	146133	0.93	808	20654	608	15542
	605	51	1.03	0.30	559	14289	0.107	133542	0.49	970	24796	814	20808
	606	54	1.23	0.42	334	8538	0.111	76919	0.59	877	22418	651	16641
July 26	607	66	2.40	0.89	431	11017	0.060	183617	1.15	826	21115	654	16718
	608	63	2.06	0.76	333	8512	0.061	139541	0.89	705	18021	534	13650
	609	65	2.06	0.74	367	9381	0.105	89343	0.98	881	22520	660	16871
	610	63	2.19	0.77	482	12321	0.063	195571	0.94	991	25332	762	19479
	611	64	2.26	0.76	292	7464	0.052	143538	1.04	527	13471	428	10941
	612	64	2.38	0.91	336	8589	0.070	122700	1.05	824	21063	597	15261
	613	61	1.79	0.60	245	6263	0.061	102672	0.83	508	12986	398	10174
	614	62	1.81	0.64	286	7311	0.058	126052	0.89	798	20399	584	14928
	615	60	1.69	0.65	309	7899	0.052	151904	0.78	665	16999	503	12858
	616	65	2.15	0.78	209	5343	0.060	89050	0.99	552	14110	401	10251
Aug 2	617	65	2.12	0.74	217	5547	0.061	90934	1.05	504	12883	385	9842
	618	64	2.42	0.90	228	5828	0.057	102246	1.19	482	12321	373	9535
	619	61	1.92	0.72	186	4755	0.049	97041	0.93	562	14366	398	10174
	620	66	2.43	0.89	275	7030	0.072	97639	1.18	527	13471	419	10711
	621	62	1.93	0.74	185	4729	0.051	92725	0.90	619	15823	423	10813
	622	72	3.48	1.38	189	4831	0.069	70014	1.73	439	11222	328	8384
	623	68	2.91	1.18	185	4729	0.049	96510	1.39	505	12909	358	9151
	624	68	1.83	0.39	282	7209	0.148	48709	0.96	508	12986	443	11324
	625	63	2.28	0.86	244	6237	0.044	141750	1.07	540	13804	408	10429
	626	62	2.05	0.69	328	8384	0.062	135226	1.00	661	16897	525	13420
Aug 9	627	60	1.77	0.65	385	9842	0.079	124582	0.89	833	21293	644	16462
	628	65	2.38	0.91	331	8461	0.055	153836	1.15	679	17357	525	13420
	629	69	2.55	1.01	198	5061	0.046	110022	1.33	452	11554	342	8742
	630	71	3.11	1.24	296	7566	0.082	92268	1.38	569	14545	440	11247
	631	61	1.92	0.73	316	8078	0.052	155346	1.01	757	19351	572	14622
	632	62	2.06	0.80	265	6774	0.043	157535	0.97	497	12704	392	10020
	633	61	1.37	0.17	504	12883	0.119	108261	0.74	473	12091	479	12244
	634	69	2.90	1.11	101	2582	0.063	40984	1.38	287	7336	204	5215
	635	69	2.90	1.20	115	2940	0.058	50690	1.33	313	8001	219	5598
	636	64	2.23	0.82	261	6672	0.054	123556	1.08	447	11426	367	9381
Aug 16	637	65	2.37	0.86	519	13267	0.048	276396	1.21	729	18635	642	16411
	638	67	2.85	1.07	388	9918	0.081	122444	1.39	619	15823	519	13267
	639	65	2.51	0.93	525	13420	0.077	174286	1.21	596	15235	565	14443
	640	63	2.09	0.77	354	9049	0.069	131145	1.00	615	15721	501	12807
	641	63	2.37	0.89	371	9484	0.046	206174	1.10	803	20527	610	15593
	642	64	2.24	0.92	466	11912	0.042	283619	1.04	603	15414	539	13778
	643	71	3.32	1.29	861	22009	0.100	220090	1.54	710	18149	779	19913
	644	59	1.76	0.66	413	10557	0.054	195500	0.83	533	13625	480	12270
	645	72	3.61	1.39	341	8717	0.057	152930	1.73	496	12679	427	10915
	646	59	1.76	0.69	577	14749	0.060	245817	0.80	839	21447	718	18354
Aug 23	647	62	2.14	0.69	585	14954	0.071	210620	1.11	581	14852	583	14903
	648	67	2.57	0.95	729	18635	0.091	204780	1.25	765	19555	749	19146
	649	70	3.36	1.26	357	9126	0.057	160105	1.65	457	11682	414	10583
	650	67	2.46	0.82	624	15951	0.064	249234	1.18	598	15286	609	15567
	651	70	3.08	1.16	563	14392	0.066	218061	1.42	675	17255	625	15976
	652	68	2.70	1.00	497	12704	0.062	204903	1.30	641	16385	578	14775
	653	76	4.02	1.63	399	10199	0.049	208143	1.92	440	11247	421	10762
	654	64	2.24	0.76	674	17229	0.095	181358	1.06	654	16718	662	16922
	655	67	2.67	0.95	512	13088	0.057	229614	1.33	540	13804	528	13497
	656	68	3.11	1.02	1117	28553	0.105	271933	1.36	790	20194	930	23773

Table A.3.5R. Cont'd.

		Whole fish		Muscle tissue					Carcass			Combined muscle/carcass	
Date(s) sampled	Fish no.	Total length (mm)	Round wt (g)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	Total Hg (μg/g)	Specific activity (dpm/μg)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)
Aug 29	657	69	2.69	0.98	525	13420	0.082	163659	1.35	704	17996	629	16079
	658	68	2.45	0.87	390	9969	0.062	160790	1.27	502	12832	456	11656
	659	57	2.32	0.90	839	21447	0.077	278532	1.11	815	20833	826	21115
Sept 30	660	75	4.00	1.52	1254	32055	0.087	368448	1.92	965	24668	1093	27940
	661	80	4.76	1.70	751	19197	0.070	274243	2.37	575	14698	649	16590
	662	81	4.86	1.73	1273	32541	0.082	396841	2.08	989	25281	1118	28579
	663	71	3.31	1.26	969	24770	0.065	381077	1.43	845	21600	903	23083
	664	77	4.37	1.58	1749	44709	0.148	302088	2.07	1194	30521	1434	36656
	665	88	6.47	2.36	1394	35634	0.095	375095	2.81	958	24489	1157	29576
	666	90	7.10	2.64	833	21293	0.146	145842	3.35	578	14775	690	17638
	667	87	6.13	2.23	672	17178	0.083	206964	2.90	528	13497	591	15107
	668	75	3.99	1.37	1509	38574	0.074	521270	1.94	1111	28400	1276	32618
	669	73	3.64	1.03	1378	35225	0.089	395787	1.81	992	25358	1132	28937

Table A.3.59. Mean mercury content of yellow perch from CHg84-6, a sediment-bottomed control limnocorral with both "cold" (ionic) mercury and "hot" ^{203}Hg, 1984. (For sample sizes, see Table A.3.58).

Date(s) sampled	Fish no.	Whole fish		Muscle tissue					Carcass			Combined muscle/carcass	
		Total length (mm)	Round wt (g)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	Total Hg (μg/g)	Specific activity (dpm/μg)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)
June 25	Time 0	59	1.72	0.66			0.073						
July 12	601-606	58	1.64	0.58	366	9347	0.082	117670	0.83	893	22827	678	17327
July 26	607-616	63	2.08	0.75	329	8410	0.064	134399	0.95	728	18602	552	14113
Aug 2	617-626	65	2.34	0.85	232	5928	0.066	97279	1.14	535	13668	406	10378
Aug 9	627-636	65	2.32	0.86	277	7086	0.065	111708	1.13	531	13566	418	10695
Aug 16	637-646	65	2.49	0.95	482	12308	0.063	200840	1.19	654	16726	578	14775
Aug 23	647-656	68	2.84	1.02	606	15483	0.072	213875	1.36	614	15698	610	15590
Aug 29	657-659	65	2.49	0.92	585	14945	0.074	200994	1.24	674	17220	637	16283
Sept 30	660-669	80	4.86	1.74	1178	30118	0.094	336766	2.27	874	22329	1004	25672

Table A.3.60. Mercury content of yellow perch from C84-7, a sediment-bottomed control limnocorral, 1984.

		Whole fish		Muscle tissue					Carcass			Combined muscle/carcass	
Date(s) sampled	Fish no.	Total length (mm)	Round wt (g)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	Total Hg (µg/g)	Specific activity (dpm/µg)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)
July 12	701	55	1.45	0.53	304	7771	0.073	106452	0.73	595	15210	473	12091
	702	56	1.59	0.44	299	7643	0.077	99260	0.95	647	16539	537	13727
	703	63	2.36	0.88	202	5164	0.091	56747	1.28	693	17715	493	12602
	704	61	2.22	0.84	212	5419	0.078	69474	1.10	482	12321	365	9330
	705	58	1.61	0.54	234	5982	0.094	63638	0.82	566	14468	434	11094
	706	61	2.04	0.77	380	9714	0.059	164644	0.91	795	20322	605	15465
July 26	707	63	2.10	0.78	288	7362	0.054	136333	0.97	609	15567	466	11912
	708	64	2.08	0.75	508	12986	0.069	188203	0.86	835	21345	683	17459
	709	56	1.36	0.44	597	15261	0.061	250180	0.67	1027	26253	857	21907
	710	65	2.22	0.77	370	9458	0.068	139088	1.08	582	14877	494	12628
	711	62	1.88	0.62	285	7285	0.063	115635	0.92	563	14392	451	11529
	712	64	2.21	0.75	285	7285	0.059	123475	1.08	484	12372	402	10276
	713	66	2.39	0.86	305	7797	0.071	109817	1.11	624	15951	485	12398
	714	63	2.22	0.80	370	9458	0.086	109977	0.98	591	15107	492	12577
	715	65	2.24	0.80	263	6723	0.071	94690	1.11	581	14852	448	11452
	716	62	1.71	0.60	553	14136	0.069	204870	0.85	762	19479	676	17280
Aug 2	717	64	2.35	0.87	365	9330	0.048	194375	1.16	670	17127	539	13778
	718	67	2.38	0.90	226	5777	0.053	109000	1.13	447	11426	349	8921
	719	58	1.55	0.58	266	6800	0.059	115254	0.66	553	14136	419	10711
	720	54	1.08	0.30	491	12551	0.092	136424	0.57	517	13216	508	12986
	721	64	2.02	0.79	333	8512	0.056	152000	0.97	667	17050	517	13216
	722	64	2.51	0.94	331	8461	0.042	201452	1.07	545	13931	445	11375
	723	64	2.24	0.83	494	12628	0.057	221544	0.98	636	16258	571	14596
	724	64	2.25	0.84	550	14059	0.071	198014	1.03	709	18124	638	16309
	725	62	1.89	0.68	430	10992	0.087	126345	0.94	571	14596	512	13088
	726	58	1.60	0.62	356	9100	0.059	154237	0.68	628	16053	498	12730
Aug 9	727	66	2.53	0.92	532	13599	0.063	215857	1.17	604	15440	572	14622
	728	80	4.72	1.76	249	6365	0.084	75774	2.29	351	8972	307	7848
	729	64	2.28	0.89	412	10532	0.046	228957	1.08	575	14698	501	12807
	730	68	2.59	1.04	337	8615	0.062	138952	1.22	513	13113	432	11043
	731	63	2.09	0.73	484	12372	0.064	193313	0.99	534	13650	513	13113
	732	60	1.64	0.59	491	12551	0.063	199222	0.75	611	15619	558	14264
	733	74	3.75	1.51	612	15644	0.066	237030	1.81	688	17587	653	16692
	734	66	2.66	0.99	660	16871	0.076	221987	1.21	703	17970	684	17485
	735	59	1.59	0.45	515	13165	0.068	193603	0.88	645	16488	601	15363
	736	65	2.56	0.93	277	7081	0.059	120017	1.18	480	12270	391	9995
Aug 16	737	66	2.44	0.91	582	14877	0.042	354214	1.25	611	15619	599	15312
	738	66	2.55	0.94	650	16616	0.064	259625	1.26	646	16513	648	16564
	739	70	3.09	1.25	928	23722	0.049	484122	1.45	878	22444	901	23032
	740	68	2.67	1.00	673	17203	0.075	229373	1.25	643	16437	656	16769
	741	62	2.34	0.93	565	14443	0.047	307298	1.06	573	14647	569	14545
	742	78	4.60	1.87	489	12500	0.038	328947	2.07	536	13701	514	13139
	743	75	4.69	1.90	785	20066	0.031	647290	1.87	636	16258	711	18175
	744	63	2.15	0.88	459	11733	0.029	404586	1.01	551	14085	508	12986
	745	70	2.87	1.22	934	23875	0.067	356343	1.22	844	21575	889	22725
	746	71	3.49	1.34	770	19683	0.050	393660	1.53	733	18737	750	19172
Aug 23	747	74	3.91	1.46	1405	35915	0.076	472566	1.91	1054	26943	1206	30828
	748	66	2.46	0.88	785	20066	0.056	358321	1.20	667	17050	717	18328
	749	72	3.18	1.19	1010	25818	0.066	391182	1.55	893	22827	944	24131
	750	65	2.39	0.90	728	18609	0.068	273662	1.17	727	18584	727	18584
	751	72	3.21	1.11	750	19172	0.069	277855	1.70	552	14110	630	16104
	752	69	2.94	1.09	902	23057	0.065	354723	1.45	844	21575	869	22214
	753	67	2.64	0.97	957	24463	0.063	388302	1.30	841	21498	891	22776
	754	73	3.57	1.29	1069	27326	0.065	420400	1.75	897	22929	970	24796
	755	81	5.39	2.08	1235	31570	0.089	354719	2.54	1128	28834	1176	30061
	756	66	2.51	0.93	845	21600	0.092	234783	1.19	720	18405	775	19811

Table A.3.60. Cont'd.

Date(s) sampled	Fish no.	Whole fish		Muscle tissue					Carcass			Combined muscle/carcass	
		Total length (mm)	Round wt (g)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	Total Hg (µg/g)	Specific activity (dpm/µg)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)
Aug 29		No sample											
Sept 30	757	70	3.00	1.11	1268	32413	0.070	463043	1.36	860	21984	1043	26662
	758	80	4.98	1.82	1159	29627	0.048	617229	2.33	848	21677	984	25153
	759	85	5.76	2.09	897	22929	0.050	458580	2.71	646	16513	755	19300
	760	74	3.39	1.23	1518	38804	0.066	587939	1.59	892	22802	1165	29780
	761	75	3.89	1.43	821	20987	0.049	428306	1.69	544	13906	671	17152
	762	90	7.14	2.54	716	18303	0.055	332782	3.48	500	12781	591	15107
	763	71	2.94	1.02	1284	32822	0.064	512844	1.42	867	22163	1041	26610
	764	78	4.70	1.71	1409	36017	0.061	590443	2.17	952	24335	1153	29473
	765	82	4.94	1.75	965	24668	0.046	536261	2.24	664	16973	796	20348
	766	71	3.39	1.27	1457	37244	0.071	524563	1.63	906	23160	1147	29320

Table A.3.61. Mean mercury content of yellow perch from C84-7, a sediment-bottomed control limnocorral, 1984. (For sample sizes, see Table A.3.60).

Date(s) sampled	Fish no.	Whole fish		Muscle tissue					Carcass			Combined muscle/carcass	
		Total length (mm)	Round wt (g)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	Total Hg (μg/g)	Specific activity (dpm/μg)	Wt (g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)	^{203}Hg (cpm/g)	^{203}Hg (dpm/g)
June 25	Time 0	59	1.72	0.66			0.073						
July 12	701-706	59	1.88	0.67	272	6949	0.079	93369	0.97	630	16096	485	12385
July 26	707-716	63	2.04	0.72	382	9775	0.067	147227	0.96	666	17020	545	13942
Aug 2	717-726	62	1.99	0.74	384	9821	0.062	160865	0.92	594	15192	500	12771
Aug 9	727-736	67	2.64	0.98	457	11680	0.065	182471	1.26	570	14581	521	13323
Aug 16	737-746	69	3.09	1.22	684	17472	0.049	376546	1.40	665	17002	675	17242
Aug 23	747-756	71	3.22	1.19	969	24760	0.071	352651	1.58	832	21276	891	22763
Aug 29		No sample											
Sept 30	757-766	78	4.41	1.6	1149	29381	0.058	505199	2.06	768	19629	935	23891

Appendix 4

Activity of radioisotopes in invertebrates, on limnocorral material, and in sediments, 1981-1983

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Appendix Table A.4.1. ^{203}Hg activity in net plankton, 1981 137

A.4.2. ^{203}Hg activity in net plankton, 1982 138

A.4.3. ^{203}Hg activity in net plankton, 1983 139

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A.4.12. Distribution of ^{203}Hg activity with depth in the limnocorral sediments, 1982 148

Table A.4.1. ^{203}Hg in netplankton; 1981 limnocorral experiment.

Date	Day of Year	C81-1				LM81-2				PN81-3			
		Sample wt (mg)	Standing crop (mg/L)	Activity (dpm/mg)	Activity (dpm/L)	Sample wt (mg)	Standing crop (mg/L)	Activity (dpm/mg)	Activity (dpm/L)	Sample wt (mg)	Standing crop (mg/L)	Activity (dpm/mg)	Activity (dpm/L)
July 3	184	5.14	0.066	1460	96.3	4.64	0.059	1237	73.6	6.37	0.082	1228	100.3
July 7	188	6.59	0.084	1162	98.1	5.33	0.068	1023	69.9	8.35	0.107	1935	207.2
July 19	200	12.16	0.156	245	38.1	6.38	0.082	685	56.0	5.26	0.067	1840	124.0
Aug 2	214	5.42	0.070	430	29.9	3.97	0.051	1342	68.3	5.98	0.077	1195	91.6
Aug 23	235	3.14	0.040	445	17.9	13.13	0.168	1039	174.9	8.91	0.114	879	100.4

Date	Day of year	LS81-4				C81-5				HS81-6			
		Sample wt (mg)	Standing crop (mg/L)	Activity (dpm/mg)	Activity (dpm/L)	Sample wt (mg)	Standing crop (mg/L)	Activity (dpm/mg)	Activity (dpm/L)	Sample wt (mg)	Standing crop (mg/L)	Activity (dpm/mg)	Activity (dpm/L)
July 3	184	3.72	0.048	1282	61.1	4.40	0.056	1117	63.0	7.10	0.091	640	58.3
July 7	188	6.15	0.079	1423	112.2	4.36	0.056	2210	123.5	5.70	0.073	1101	80.5
July 19	200	7.36	0.094	600	56.6	7.90	0.101	970	98.2	6.54	0.084	489	41.0
Aug 2	214	7.50	0.096	416	40.0	6.69	0.086	1431	122.7	7.54	0.097	444	42.9
Aug 23	235	7.88	0.101	371	37.5	3.07	0.039	2015	79.3	6.32	0.081	369	29.9

Date	Day of year	C81-7			
		Sample wt (mg)	Standing crop (mg/L)	Activity (dpm/mg)	Activity (dpm/L)
July 3	184	6.88	0.088	1432	126.3
July 7	188	6.12	0.078	2190	171.8
July 19	200	8.33	0.107	1829	195.3
Aug 2	214	3.83	0.049	1396	68.5
Aug 23	235	19.17	0.246	970	238.4

Table A.4.2. ^{203}Hg in netplankton; 1982 limnocorral experiment.

Date	Day of year	FM82-1 Sample wt (mg)	FM82-1 Standing crop (mg/L)	FM82-1 Activity (dpm/mg)	FM82-1 Activity (dpm/L)	LM82-3 Sample wt (mg)	LM82-3 Standing crop (mg/L)	LM82-3 Activity (dpm/mg)	LM82-3 Activity (dpm/L)	HM82-4 Sample wt (mg)	HM82-4 Standing crop (mg/L)	HM82-4 Activity (dpm/mg)	HM82-4 Activity (dpm/L)
July 5	186	5.85	0.078	714	55.4	4.91	0.065	356	23.2	2.27	0.030	926	27.9
July 12	193	-	-	-	-	2.40	0.032	604	19.2	4.97	0.066	290	19.1
July 14	195	7.99	0.106	405	42.9	-	-	-	-	-	-	-	-
July 26	207	3.57	0.047	494	23.4	2.59	0.034	404	13.9	3.31	0.044	365	16.0
Aug 2	214	10.85	0.144	177	25.4	-	-	-	-	-	-	-	-
Aug 4	216	25.71	0.341	31	10.0	9.42	0.125	146	18.3	23.63	0.313	86	26.9
Aug 9	221	18.69	0.248	83	20.7	40.57	0.538	197	105.8	18.99	0.252	139	34.9
Aug 16	228	0.78	0.010	1328	13.7	11.43	0.152	49	7.5	13.50	0.179	152	27.2
Aug 23	235	3.66	0.049	207	10.1	2.67	0.035	240	8.5	6.10	0.081	162	13.1
Aug 28	240	9.65	0.128	99	12.7	7.93	0.105	46	4.8	18.50	0.245	54	13.2

Date	Day of year	OC82-5 Sample wt (mg)	OC82-5 Standing crop (mg/L)	OC82-5 Activity (dpm/mg)	OC82-5 Activity (dpm/L)	C82-6 Sample wt (mg)	C82-6 Standing crop (mg/L)	C82-6 Activity (dpm/mg)	C82-6 Activity (dpm/L)	HS82-7 Sample wt (mg)	HS82-7 Standing crop (mg/L)	HS82-7 Activity (dpm/mg)	HS82-7 Activity (dpm/L)
July 5	186	5.38	0.071	463	33.0	4.96	0.066	1096	72.1	19.71	0.261	179	46.8
July 12	193	-	-	-	-	5.27	0.070	925	64.7	9.45	0.125	331	41.4
July 14	195	11.54	0.153	73	11.2	-	-	-	-	-	-	-	-
July 26	207	0.58	0.008	1331	10.2	4.53	0.060	321	19.3	5.45	0.072	186	13.5
Aug 2	214	-	-	-	-	-	-	-	-	-	-	-	-
Aug 4	216	2.94	0.039	206	8.1	6.90	0.092	172	15.8	6.04	0.080	84	6.9
Aug 9	221	3.15	0.042	297	12.4	7.90	0.105	135	14.2	9.88	0.131	80	10.5
Aug 16	228	7.68	0.102	91	9.3	7.93	0.105	89	9.3	3.34	0.044	141	6.2
Aug 23	235	4.52	0.060	187	11.2	20.50	0.272	87	23.7	3.54	0.047	87	4.1
Aug 28	240	18.15	0.241	60	14.4	12.93	0.171	25	6.1	6.97	0.092	25	2.3

Date	Day of year	OM82-8 Sample wt (mg)	OM82-8 Standing crop (mg/L)	OM82-8 Activity (dpm/mg)	OM82-8 Activity (dpm/L)	L82-9 Sample wt (mg)	L82-9 Standing crop (mg/L)	L82-9 Activity (dpm/mg)	L82-9 Activity (dpm/L)
July 5	186	3.06	0.041	784	31.8	-	-	-	-
July 12	193	2.94	0.039	736	28.7	-	-	-	-
July 14	195	-	-	-	-	-	-	-	-
July 26	207	20.72	0.275	270	74.1	1.74	0.023	-	-
Aug 2	214	-	-	-	-	-	-	-	-
Aug 4	216	34.84	0.462	203	93.9	7.32	0.097	-	-
Aug 9	221	17.85	0.237	218	51.6	3.43	0.046	-	-
Aug 16	228	6.80	0.090	435	39.2	3.99	0.053	-	-
Aug 23	235	2.94	0.039	410	16.0	2.79	0.037	-	-
Aug 28	240	8.00	0.106	173	18.3	3.59	0.048	-	-

Table A.4.3. ^{203}Hg in netplankton; 1983 limnocorral experiment.

Date	Day of year	DFM83-1				SP83-2				LM83-3			
		Sample wt (mg)	Standing crop (mg/L)	Activity (dpm/mg)	Activity (dpm/L)	Sample wt (mg)	Standing crop (mg/L)	Activity (dpm/mg)	Activity (dpm/L)	Sample wt (mg)	Standing crop (mg/L)	Activity (dpm/mg)	Activity (dpm/L)
July 8	189	5.30	0.070	1182	83.1	1.60	0.021	2873	61.0	4.40	0.058	848	49.5
July 15	196	3.75	0.050	1586	78.9	1.54	0.020	4285	87.5	1.44	0.019	4218	80.6
July 21	202	9.33	0.124	1249	154.6	1.82	0.024	1694	40.9	5.76	0.076	1301	99.4
July 29	210	9.65	0.128	593	75.9	2.94	0.039	1140	44.5	2.06	0.027	1013	27.7
Aug 5	217	-	-	-	-	-	-	-	-	1.78	0.024	600	14.2
Aug 12	224	24.42	0.324	221	71.6	9.30	0.123	101	12.5	8.40	0.111	388	43.2
Aug 20	232	11.11	0.147	256	37.7	2.66	0.035	153	5.4	8.15	0.108	255	27.6
Aug 25	237	-	-	-	-	-	-	-	-	8.86	0.118	187	22.0
Aug 26	238	22.84	0.303	144	43.6	9.06	0.120	94	11.3	2.62	0.035	430	14.9

Date	Day of year	C83-4				OAS83-5				OM83-6			
		Sample wtht (mg)	Standing crop (mg/L)	Activity (dpm/mg)	Activity (dpm/L)	Sample wt (mg)	Standing crop (mg/L)	Activity (dpm/mg)	Activity (dpm/L)	Sample wt (mg)	Standing crop (mg/L)	Activity (dpm/mg)	Activity (dpm/L)
July 8	189	6.50	0.086	626	54.0	1.90	0.025	2451	61.8	12.20	0.162	642	103.9
July 15	196	0.65	0.009	6183	53.3	1.75	0.023	3128	72.6	2.21	0.029	2806	82.2
July 21	202	1.74	0.023	1462	33.7	3.15	0.042	1190	49.7	7.39	0.098	531	52.0
July 29	210	3.78	0.050	703	35.2	2.82	0.037	1102	41.2	0.30	0.004	6867	27.3
Aug 5	217	5.83	0.077	335	25.9	2.70	0.036	375	13.4	9.32	0.124	501	61.9
Aug 12	224	6.67	0.088	320	28.3	9.89	0.131	696	91.3	8.77	0.116	491	57.1
Aug 20	232	19.08	0.253	236	59.7	11.27	0.149	249	37.2	7.65	0.101	367	37.2
Aug 25	237	11.01	0.146	171	25.0	16.99	0.225	126	28.4	7.21	0.096	296	28.3
Aug 26	238	-	-	-	-	-	-	-	-	-	-	-	-

Date	Day of year	LM83-7				LMS83-8				L83-9			
		Sample wt (mg)	Standing crop (mg/L)	Activity (dpm/mg)	Activity (dpm/L)	Sample wt (mg)	Standing crop (mg/L)	Activity (dpm/mg)	Activity (dpm/L)	Sample wt (mg)	Standing crop (mg/L)	Activity (dpm/mg)	Activity (dpm/L)
July 8	189	9.90	0.131	454	59.6	3.20	0.042	1132	48.0	3.00	0.040	-	-
July 15	196	3.34	0.044	1989	88.1	5.25	0.070	928	64.6	1.80	0.024	-	-
July 21	202	15.07	0.200	683	136.5	6.07	0.081	466	37.5	1.91	0.025	-	-
July 29	210	2.72	0.036	945	34.1	3.87	0.051	304	15.6	-	-	-	-
Aug 5	217	3.77	0.050	469	23.5	6.84	0.091	162	14.7	7.50	0.099	-	-
Aug 12	224	6.37	0.084	443	37.4	3.27	0.043	318	13.8	1.10	0.015	-	-
Aug 20	232	24.73	0.328	90	29.5	24.73	0.328	230	75.4	1.87	0.025	-	-
Aug 25	237	11.95	0.158	180	28.5	11.95	0.158	136	21.6	-	-	-	-
Aug 26	238	-	-	-	-	-	-	-	-	2.45	0.032	-	-

Table A.4.4. ^{75}Se in netplankton; 1981 limnocorral experiment.

Date	Day of year	C81-1				LM81-2				PN81-3			
		Sample wt (mg)	Standing crop (mg/L)	Activity (dpm/mg)	Activity (dpm/L)	Sample wt (mg)	Standing crop (mg/L)	Activity (dpm/mg)	Activity (dpm/L)	Sample wt (mg)	Standing crop (mg/L)	Activity (dpm/mg)	Activity (dpm/L)
July 3	184	5.14	0.066	791	52.2	4.64	0.059	610	36.3	6.37	0.082	732	59.8
July 7	188	6.59	0.084	1015	85.7	5.33	0.068	786	53.7	8.35	0.107	817	87.5
July 19	200	12.16	0.156	212	33.1	6.38	0.082	610	49.9	5.26	0.067	818	55.2
Aug 2	214	5.42	0.070	237	16.5	3.97	0.051	766	39.0	5.98	0.077	854	65.4
Aug 23	235	3.14	0.040	507	20.4	13.13	0.168	305	51.3	8.91	0.114	571	65.3

Date	Day of year	LS81-4				C81-5				HS81-6			
		Sample wt (mg)	Standing crop (mg/L)	Activity (dpm/mg)	Activity (dpm/L)	Sample wt (mg)	Standing crop (mg/L)	Activity (dpm/mg)	Activity (dpm/L)	Sample wt (mg)	Standing crop (mg/L)	Activity (dpm/mg)	Activity (dpm/L)
July 3	184	3.72	0.048	467	22.3	4.40	0.056	366	20.7	7.10	0.091	230	20.9
July 7	188	6.15	0.079	742	58.5	4.36	0.056	1096	61.3	5.70	0.073	875	63.9
July 19	200	7.36	0.094	220	20.8	7.90	0.101	329	33.3	6.54	0.084	270	22.6
Aug 2	214	7.50	0.096	518	49.8	6.69	0.086	1266	108.6	7.54	0.097	796	77.0
Aug 23	235	7.88	0.101	814	82.2	3.07	0.039	-	-	6.32	0.081	445	36.1

Date	Day of year	C81-7			
		Sample wt (mg)	Standing crop (mg/L)	Activity (dpm/mg)	Activity (dpm/L)
July 3	184	6.88	0.088	526	46.4
July 7	188	6.12	0.078	1277	100.2
July 19	200	8.33	0.107	661	70.6
Aug 2	214	3.83	0.049	909	44.7
Aug 23	235	19.17	0.246	442	108.7

Table A.4.5. ^{234}Cs in netplankton; 1981 limnocorral experiment.

Date	Day of year	C81-1				LM81-2				PN81-3			
		Sample wt (mg)	Standing crop (mg/L)	Activity (dpm/mg)	Activity (dpm/L)	Sample wt (mg)	Standing crop (mg/L)	Activity (dpm/mg)	Activity (dpm/L)	Sample wt (mg)	Standing crop (mg/L)	Activity (dpm/mg)	Activity (dpm/L)
July 3	184	5.14	0.066	193	12.7	4.64	0.059	153	9.1	6.37	0.082	100	8.2
July 7	188	6.59	0.084	174	14.7	5.33	0.068	152	10.4	8.35	0.107	155	16.6
July 19	200	12.16	0.156	44	6.8	6.38	0.082	45	3.7	5.26	0.067	120	8.1
Aug 2	214	5.42	0.070	147	10.2	3.97	0.051	136	6.9	5.98	0.077	247	18.9
Aug 23	235	3.14	0.040	105	4.2	13.13	0.168	90	15.2	8.91	0.114	203	23.2

Date	Day of year	LS81-4				C81-5				HS81-6			
		Sample wt (mg)	Standing crop (mg/L)	Activity (dpm/mg)	Activity (dpm/L)	Sample wt (mg)	Standing crop (mg/L)	Activity (dpm/mg)	Activity (dpm/L)	Sample wt (mg)	Standing crop (mg/L)	Activity (dpm/mg)	Activity (dpm/L)
July 3	184	3.72	0.048	199	9.5	4.40	0.056	87	4.9	7.10	0.091	162	14.8
July 7	188	6.15	0.079	149	11.8	3.36	0.056	173	9.7	5.70	0.073	153	11.2
July 19	200	7.36	0.094	60	5.6	7.90	0.101	88	8.9	6.54	0.084	62	5.2
Aug 2	214	7.50	0.096	96	9.2	6.69	0.086	192	16.5	7.54	0.097	68	6.6
Aug 23	235	7.88	0.101	52	5.3	3.07	0.039	394	15.5	6.32	0.081	178	14.4

Date	Day of year	C81-7			
		Sample wt (mg)	Standing crop (mg/L)	Activity (dpm/mg)	Activity (dpm/L)
July 3	184	6.88	0.088	139	12.2
July 7	188	6.12	0.078	145	11.4
July 19	200	8.33	0.107	482	51.5
Aug 2	214	3.83	0.049	391	19.2
Aug 23	235	19.17	0.246	1203	295.7

Table A.4.6. ^{60}Co in netplankton; 1981 limnocorral experiment.

Date	Day of year	C81-1				LM81-2				PN81-3			
		Sample wt (mg)	Standing crop (mg/L)	Activity (dpm/mg)	Activity (dpm/L)	Sample wt (mg)	Standing crop (mg/L)	Activity (dpm/mg)	Activity (dpm/L)	Sample wt (mg)	Standing crop (mg/L)	Activity (dpm/mg)	Activity (dpm/L)
July 3	184	5.14	0.066	2588	170.6	4.64	0.059	3601	214.2	6.37	0.032	2573	210.1
July 7	188	6.59	0.084	1751	147.8	5.33	0.068	1528	104.4	8.35	0.107	3044	325.9
July 19	200	12.16	0.156	189	29.5	6.38	0.082	350	28.7	5.26	0.067	1249	84.2
Aug 2	214	5.42	0.070	661	46.0	3.97	0.051	788	40.1	5.98	0.077	1411	108.1
Aug 23	235	3.14	0.040	792	31.9	13.13	0.168	803	135.3	8.91	0.114	952	108.8

Date	Day of year	LS81-4				C81-5				HS81-6			
		Sample wt (mg)	Standing crop (mg/L)	Activity (dpm/mg)	Activity (dpm/L)	Sample wt (mg)	Standing crop (mg/L)	Activity (dpm/mg)	Activity (dpm/L)	Sample wt (mg)	Standing crop (mg/L)	Activity (dpm/mg)	Activity (dpm/L)
July 3	184	3.72	0.048	2062	98.4	4.40	0.056	1796	101.3	7.10	0.091	1272	115.8
July 7	188	6.15	0.079	2508	197.7	4.36	0.056	5218	291.6	5.70	0.073	1916	140.0
July 19	200	7.36	0.094	767	72.4	7.90	0.101	1770	179.3	6.54	0.084	414	34.7
Aug 2	214	7.50	0.096	1088	104.6	6.69	0.086	2084	178.8	7.54	0.097	599	57.9
Aug 23	235	7.88	0.101	675	68.1	3.07	0.039	2261	89.0	6.32	0.081	688	55.8

Date	Day of year	C81-7			
		Sample wt (mg)	Standing crop (mg/L)	Activity (dpm/mg)	Activity (dpm/L)
July 3	184	6.88	0.088	2295	202.4
July 7	188	6.12	0.078	3630	284.8
July 19	200	8.33	0.107	2530	270.2
Aug 2	214	3.83	0.049	2383	117.0
Aug 23	235	19.17	0.246	4196	1031.3

Table A.4.7. ^{65}Zn in netplankton; 1981 limnocorral experiment.

Date	Day of year	C81-1 Sample wt (mg)	C81-1 Standing crop (mg/L)	C81-1 Activity (dpm/mg)	C81-1 Activity (dpm/L)	LM81-2 Sample wt (mg)	LM81-2 Standing crop (mg/L)	LM81-2 Activity (dpm/mg)	LM81-2 Activity (dpm/L)	PN81-3 Sample wt (mg)	PN81-3 Standing crop (mg/L)	PN81-3 Activity (dpm/mg)	PN81-3 Activity (dpm/L)
July 3	184	5.14	0.066	632	41.7	4.64	0.059	275	16.4	6.37	0.082	902	73.6
July 7	188	6.59	0.084	973	82.1	5.33	0.068	359	24.6	8.35	0.107	1671	178.8
July 19	200	12.16	0.156	148	23.0	6.38	0.082	248	20.3	5.26	0.067	1074	72.4
Aug 2	214	5.42	0.070	-	-	3.97	0.051	-	-	5.98	0.077	-	-
Aug 23	235	3.14	0.040	612	24.6	13.13	0.168	283	47.7	8.91	0.114	892	101.8

Date	Day of year	LS81-4 Sample wt (mg)	LS81-4 Standing crop (mg/L)	LS81-4 Activity (dpm/mg)	LS81-4 Activity (dpm/L)	C81-5 Sample wt (mg)	C81-5 Standing crop (mg/L)	C81-5 Activity (dpm/mg)	C81-5 Activity (dpm/L)	HS81-6 Sample wt (mg)	HS81-6 Standing crop (mg/L)	HS81-6 Activity (dpm/mg)	HS81-6 Activity (dpm/L)
July 3	184	3.72	0.048	646	30.8	4.40	0.056	456	25.7	7.10	0.091	310	28.2
July 7	188	6.15	0.079	1064	83.9	4.36	0.056	1424	79.6	5.70	0.073	1237	90.4
July 19	200	7.36	0.094	214	20.2	7.90	0.101	544	55.1	6.54	0.084	72	6.0
Aug 2	214	7.50	0.096	-	-	6.69	0.086	-	-	7.54	0.097	104	10.0
Aug 23	235	7.88	0.101	372	37.6	3.07	0.039	908	35.7	6.32	0.081	325	26.3

Date	Day of year	C81-7 Sample wt (mg)	C81-7 Standing crop (mg/L)	C81-7 Activity (dpm/mg)	C81-7 Activity (dpm/L)
July 3	184	6.88	0.088	-	-
July 7	188	6.12	0.078	1779	139.6
July 19	200	8.33	0.107	930	99.3
Aug 2	214	3.83	0.049	-	-
Aug 23	235	19.17	0.246	1344	330.3

Table A.4.8. ^{203}Hg in emergent insects; 1983 limnocorral experiment.

		LM83-3				C83-4		OAS83-5				OM83-6			
		Diptera		Ephemeroptera		Diptera		Diptera		Trichoptera		Diptera		Trichoptera	
Period of Emergence	Duration (days)	Sample weight (mg)	Activity (dpm/mg)	Sample weight (mg)	Activity (dpm/mg)	Sample weight (mg)	Activity (dpm/mg)	Sample weight (mg)	Activity (dpm/mg)	Sample weight (mg)	Activity (dpm/mg)	Sample weight (mg)	Activity (dpm/mg)	Sample weight (mg)	Activity (dpm/mg)
July 29 - Aug 4	6	0.98	308	6.97	21	5.15	90	13.84	7	20.99	2	19.19	104	18.65	12
Aug 15 - Aug 20	5	10.92	140	-	-	10.44	57	10.41	136	-	-	24.25	79	-	-

Table A.4.9. Isotope activity on periphyton strips (dpm/cm^2) on Aug 25 (day 237); 1981 limnocorral experiment. (The first depth interval, 0.1-0.0, is above the water surface).

Limnocorral	Depth range (m)	Isotope				
		^{203}Hg	^{75}Se	^{134}Cs	^{60}Co	^{65}Zn
C81-1	0.1 - 0.0	34.9	4.7	74.1	51.5	19.6
	0.0 - 0.1	17.4	12.5	30.1	77.3	35.7
	0.5 - 0.6	45.3	15.6	20.8	242.3	57.1
	0.9 - 1.0	118.6	29.7	60.2	675.3	76.8
	1.3 - 1.4	186.0	32.8	67.1	845.4	132.1
	1.7 - 1.8	203.5	45.3	67.1	762.9	182.1
	Means:	101.0	23.4	53.2	442.4	83.9
LM81-2	0.1 - 0.0	8.1	3.1	27.8	20.6	14.3
	0.0 - 0.1	29.1	7.8	16.2	123.7	14.3
	0.5 - 0.6	51.2	29.7	13.9	314.4	25.0
	0.9 - 1.0	75.6	14.1	16.2	329.9	35.7
	1.3 - 1.4	69.8	14.1	9.3	345.4	30.4
	1.7 - 1.8	89.5	15.6	11.6	407.2	44.6
	Means:	53.9	14.1	15.8	256.9	27.4
PN81-3	0.1 - 0.0	9.3	7.8	125.0	67.0	-
	0.0 - 0.1	37.2	10.9	76.4	108.2	-
	0.5 - 0.6	91.9	28.1	115.7	314.4	-
	0.9 - 1.0	72.1	20.3	74.1	201.0	-
	1.3 - 1.4	53.5	17.2	53.2	190.7	-
	1.7 - 1.8	73.3	17.2	57.9	170.1	-
	Means:	56.2	16.9	83.7	175.3	-
LS81-4	0.1 - 0.0	24.4	7.8	129.6	123.7	-
	0.0 - 0.1	40.7	18.8	50.9	366.0	-
	0.5 - 0.6	76.7	23.4	48.6	587.6	-
	0.9 - 1.0	65.1	32.8	34.7	541.2	-
	1.3 - 1.4	115.1	31.3	44.0	799.0	-
	1.7 - 1.8	140.7	48.4	64.8	1206.2	-
	Means:	77.1	27.1	62.1	604.0	-
C81-5	0.1 - 0.0	61.2	24.0	118.8	1034.4	145.2
	0.0 - 0.1	152.3	48.4	111.1	2412.4	250.9
	0.5 - 0.6	191.9	51.6	125.0	2824.7	269.6
	0.9 - 1.0	193.0	46.9	129.6	2752.6	260.7
	1.0 - 1.1	158.1	39.1	113.4	2288.7	204.5
	1.4 - 1.5	119.8	32.8	67.1	1701.0	142.0
	1.7 - 1.8	127.9	31.3	55.6	1814.4	135.7
	Means:	143.5	39.1	103.0	2118.3	201.2
HS81-6	0.1 - 0.0	15.1	3.1	46.3	51.5	55.4
	0.0 - 0.1	43.0	14.1	145.8	190.7	126.8
	0.5 - 0.6	57.0	23.4	27.8	443.3	32.1
	0.9 - 1.0	84.9	26.6	25.5	479.4	55.4
	1.3 - 1.4	46.5	20.3	11.6	278.4	14.3
	1.7 - 1.8	64.0	28.1	16.2	319.6	17.9
	Means:	51.7	19.3	45.5	293.8	50.3
C81-7	0.1 - 0.0	15.1	2.8	85.6	77.3	-
	0.0 - 0.1	30.2	12.5	85.6	433.0	-
	0.5 - 0.6	65.1	17.2	57.9	1866.0	-
	0.9 - 1.0	68.6	25.0	76.4	1783.5	-
	1.3 - 1.4	74.4	15.6	44.0	1505.2	-
	1.7 - 1.8	68.6	14.1	34.7	1216.5	-
	Means:	53.7	14.5	64.0	1146.9	-

Table A.4.10. ^{203}Hg activity on periphyton strips; 1983 limnocorral experiment. On Aug 11 only, replicate strips were counted.

Limnocorral	Replicate	July 29		Aug 11	
		^{203}Hg (dpm/mg)	^{203}Hg (dpm/cm^2)	^{203}Hg (dpm/mg)	^{203}Hg (dpm/cm^2)
DFM83-1	A	1706	242	250	19
	B	-	-	288	13
SP83-2	A	2693	133	-	-
LM83-3	A	1649	72	1457	283
	B	-	-	1411	303
C83-4	A	1799	37	948	125
	B	-	-	1012	125
OAS83-5	A	990	170	522	88
	B	-	-	406	60
OM83-6	A	911	20	439	5
	B	-	-	453	12
LM83-7	A	1351	26	518	60
	B	-	-	693	106
LMS83-8	A	369	128	146	213
	B	-	-	217	233

Table A.4.11. Isotope activity in sediment cores from C81-1 on Aug 26 (Day 238); 1981 limnocorral experiment.

Replicate	Depth range of core section (cm)	Isotope				
		^{203}Hg (dpm/g)	^{75}Se (dpm/g)	^{134}Cs (dpm/g)	^{60}Co (dpm/g)	^{65}Zn (dpm/g)
Core #1	0-1	8428	1759	10912	53330	8459
	1-2	298	91	310	1722	279
	2-3	40	28	32	196	45
	3-4	19	31	37	124	38
	4-5	14	13	9	52	18
	5-6	26	25	14	52	21
	6-7	7	16	9	41	21
Total core activity: (dpm/m^2)		5355103	1246076	6823868	33500458	5396488
Core #2	0-1	12888	3113	16782	85474	13677
	1-2	1384	375	1769	10093	1434
	2-3	293	97	417	2165	302
	3-4	181	100	356	2041	255
	4-5	0	22	28	206	38
	5-6	21	16	19	103	32
	6-7	16	13	14	52	20
	7-8	0	28	9	41	21
	8-9	14	22	19	62	23
	9-10	33	28	19	72	21
	10-11	12	6	0	21	13
	11-12	9	6	5	21	11
Total core activity: (dpm/m^2)		117041	122813	85442	389977	155308
Core #3	0-1	15740	3819	21523	102495	18036
	1-2	665	244	907	5175	827
	2-3	233	53	171	887	171
	3-4	53	59	88	474	105
	4-5	35	9	23	113	34
	5-6	16	25	37	82	46
	6-7	0	25	19	93	39
	7-8	16	28	37	62	41
	8-9	0	16	19	52	25
	9-10	49	34	32	72	73
	11-12	0	6	9	10	11
	12-13	16	9	14	21	14
	13-14	5	3	5	10	9
	14-15	14	16	14	31	25
Total core activity: (dpm/m^2)		124793	135208	161317	303780	232738

Table A.4.12. ^{203}Hg activity in sediment cores (dpm/g); 1982 limnocorral experiment.

Limnocorral	Replicate	Depth range of core section (cm)								Total activity in core (dpm/m^2)
		0-1	1-2	2-3	3-4	4-5	5-6	6-7	7-8	
FM82-1	Core #2	34674	24070	10760	10367	1067	312	-	-	13089471
LM82-3	Core #1	11840	1912	181	105	-	-	-	-	6789170
	Core #2	14570	2637	972	1388	-	-	-	-	8982265
HM82-4	Core #1	75558	15451	2356	1181	-	-	-	-	6536076
	Core #2	6337	1295	847	1088	-	-	-	-	4869024
OC82-5	Core #1	14449	1335	765	72	98	0	0	0	7317751
	Core #2	19091	1249	500	79	112	0	-	-	10154151
C82-6	Core #1	10549	4735	1979	723	-	-	-	-	7166683
	Core #2	39677	7895	1388	405	-	-	-	-	15502016
HS82-7	Core #1	10298	4528	1765	428	-	-	-	-	12046487
	Core #2	19556	15872	1021	242	-	-	-	-	21863578
OM82-8	Core #1	42163	7640	4323	1170	4153	2021	165	-	27964357
	Core #2	26988	3970	272	200	240	107	0	-	12855563

Appendix 5

Summary of water chemistry in the limnocorrals, 1981-1983

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A.5.5. *In situ* respiration in the limnocorrals through the night of Aug 19-20, 1981 158

Table A.5.1. Chemistry data for 1981 limnocorrals. (Units are μmoles/L unless otherwise stated).

Limnocorral	Sample no.	Day of year	Cl	SO_4	Na	K	Ca	Mg	pH	DIC	Spec Cond μS/cm	Susp N	TDN	NO_3	NO_2	NH_4	Susp P	TDP	SRP	DOC	Susp C	SRSi	Fe	Mn	Chl a μg/L	TSS mg/L	O_2	O_2	
																												% Sat'n	Resp'n μmol/m^3-h
C81-1	36	171	25	40	91	21.9	219	106	-	720	80	2.8	23.6	<0.07	-	0.20	0.26	0.16	-	750	31.8	29.1	0.75	<0.36	1.2	3	-	-	-
	44	174	-	-	-	-	-	-	-	836	-	-	-	-	-	-	-	-	-	-	-	-	-	-	1.6	4	-	-	-
	52	183	17	39	101	25.7	238	119	-	688	80	7.8	20.7	0.11	<0.01	0.61	0.31	0.26	0.03	650	47.0	22.7	1.10	<0.36	3.6	4	293	-	280
	68	191	20	39	104	26.9	243	117	8.05	823	82	12.9	20.0	0.04	<0.01	0.30	0.35	0.19	<0.03	630	71.4	22.9	1.10	<0.36	4.9	5	291	96	303
	85	201	23	39	102	25.7	232	110	7.99	726	86	11.9	22.8	0.03		0.53	0.27	0.19	0.03	1900	62.6	21.5	1.10	<0.36	4.4	3	293	100	496
	176	215	17	41	100	23.8	243	117	8.23	823	83	9.9	21.4	<0.01	<0.01	0.36	0.27	0.26	<0.03	680	65.3	18.6	0.44	<0.18	4.5	3	299	100	439
	217	232	-	-	-	-	-	-	-	-	-	11.0	-	-	-	-	0.44	-	-	-	47.5	-	-	-	2.4	-	282	96	243
	230	236	23	40	106	22.6	274	144	8.17	748	91	5.1	27.8	0.03	0.01	0.31	0.14	0.26	<0.03	800	37.5	12.1	<0.36	<0.18	2.4	3	284	101	194
LM81-2	37	171	45	144	89	21.3	230	114	-	662	86	2.9	24.3	<0.07	-	0.30	0.25	0.16	-	1620	29.8	28.8	0.38	<0.36	1.2	3	-	-	-
	45	175	-	-	-	-	-	-	-	653	-	-	-	-	-	-	-	-	-	-	-	-	-	-	3.0	3	-	-	-
	53	183	14	39	103	25.7	174	98	-	620	66	8.8	21.4	0.13	<0.01	0.53	0.48	0.26	<0.03	690	58.6	23.0	2.75	<0.36	5.2	4	256	-	360
	69	191	20	40	103	29.0	177	100	7.56	675	64	10.3	22.8	0.04	<0.01	0.17	0.40	0.26	0.04	700	54.0	25.0	1.10	<0.36	5.4	5	245	80	187
	86	201	23	39	98	29.0	174	94	7.29	608	-	6.8	26.4	0.04	-	0.45	0.41	0.29	0.05	760	45.3	22.2	1.65	<0.36	3.0	-	226	78	625
	177	215	17	39	101	28.3	190	108	7.73	796	70	4.0	30.0	<0.01	0.01	0.18	0.25	0.58	0.11	810	26.3	18.0	0.88	<0.18	1.7	-	240	81	366
	218	232	-	-	-	-	-	-	-	-	-	5.4	-	-	-	-	0.82	-	-	-	49.0	-	-	-	2.6	-	244	83	408
	231	236	20	37	107	32.8	209	133	7.44	726	75	7.6	29.3	0.04	0.02	0.17	0.39	0.71	0.29	840	60.6	10.9	1.32	<0.18	2.7	2	232	83	358
PN81-3	38	171	21	213	89	20.8	235	116	-	679	97	2.6	22.1	0.07	-	0.11	0.23	0.16	-	1720	26.0	30.5	1.13	<0.36	1.0	3	-	-	-
	46	174	-	-	-	-	-	-	-	784	-	-	-	-	-	-	-	-	-	-	-	-	-	-	2.4	3	-	-	-
	54	183	14	39	104	25.2	224	112	-	648	83	11.9	22.8	0.73	0.02	0.32	0.40	0.26	0.03	650	58.3	25.6	1.10	<0.36	6.7	3	305	-	333
	65	189	-	-	-	-	-	-	-	675	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	70	191	20	39	100	24.4	211	106	8.95	659	82	26.7	21.4	0.04	<0.01	0.23	0.44	0.19	0.04	600	141.0	24.6	0.55	<0.36	13.7	5	319	105	620
	87	201	20	39	103	22.3	211	106	9.42	547	80	31.4	25.7	0.04	-	0.37	0.48	0.16	0.07	710	162.0	22.8	1.10	<0.36	14.4	4	333	114	1274
	178	215	17	40	103	20.9	214	108	8.98	670	78	24.6	30.0	<0.01	<0.01	0.23	0.50	0.29	<0.03	720	158.9	16.6	<0.36	<0.18	10.9	-	304	102	858
	190	226	-	-	-	-	-	-	-	596	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	200	229	-	-	-	-	-	-	-	609	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	219	232	-	-	-	-	-	-	9.05	647	-	27.3	-	-	-	-	0.61	-	-	-	188.5	-	-	-	9.6	-	296	102	383
	232	236	20	41	110	21.5	232	139	9.28	609	80	21.6	35.0	0.04	0.01	0.31	0.52	0.29	<0.03	860	165.4	9.8	0.88	<0.18	9.4	4	305	109	801
LS81-4	39	171	23	162	89	21.9	240	117	-	668	85	2.2	22.8	0.07	-	0.16	0.23	0.16	-	1620	25.1	30.4	0.75	<0.36	1.1	3	-	-	-
	47	175	-	-	-	-	-	-	-	672	-	-	-	-	-	-	-	-	-	-	-	-	-	-	2.1	11	-	-	-
	55	183	11	39	102	25.7	243	112	-	754	90	5.8	20.7	0.07	<0.01	0.41	0.34	0.23	0.04	590	38.0	27.6	1.10	<0.36	4.1	9	292	-	205
	71	191	23	40	98	24.8	251	110	8.31	900	90	9.6	20.0	0.03	<0.01	0.34	0.48	0.26	0.07	570	61.3	28.2	1.10	<0.36	7.6	12	293	96	289
	88	201	20	39	101	24.4	266	110	8.43	727	95	11.0	20.7	0.02	-	0.29	0.36	0.16	0.04	610	71.1	27.1	1.10	<0.36	6.2	-	287	99	455
	179	215	17	40	100	23.2	290	121	8.49	1060	95	11.4	23.6	<0.01	<0.01	0.20	0.49	0.29	0.04	660	75.6	24.8	2.21	<0.18	4.7	13	288	96	389
	191	226	-	-	-	-	-	-	-	925	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	201	229	-	-	-	-	-	-	-	903	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	220	232	-	-	-	-	-	-	8.07	1015	-	8.7	-	-	-	-	0.34	-	-	-	58.0	-	-	-	1.8	-	272	93	70
	233	236	23	42	105	23.8	333	137	8.22	1020	106	9.5	26.4	0.35	0.15	1.26	0.30	0.26	0.03	700	53.6	22.3	0.88	<0.18	1.8	10	268	96	270
C81-5	40	171	28	187	89	20.8	230	112	-	664	93	3.0	22.1	<0.07	-	0.12	0.26	0.13	-	1950	25.6	30.6	0.75	<0.36	0.9	3	-	-	-
	48	174	-	-	-	-	-	-	-	766	-	-	-	-	-	-	-	-	-	-	-	-	-	-	1.4	-	-	-	-
	56	183	17	40	103	24.4	219	110	-	725	78	7.6	18.6	0.08	<0.01	0.11	0.23	0.23	<0.03	570	39.6	27.6	1.10	<0.36	3.6	3	299	-	281
	72	191	25	42	101	25.2	214	104	8.27	760	78	13.9	20.7	0.03	<0.01	0.21	0.30	0.19	0.05	600	75.3	27.5	0.55	<0.36	6.8	3	300	98	382
	89	201	20	39	98	23.2	214	110	8.29	736	83	14.4	22.8	0.02		0.33	0.26	0.16	0.04	610	82.6	24.6	1.10	<0.36	5.3	-	302	104	504
	180	215	17	40	99	20.9	222	112	8.21	867	77	10.9	30.7	<0.01	<0.01	0.11	0.26	0.19	<0.03	690	70.8	15.9	<0.36	<0.18	3.4	2	296	99	394
	192	226	-	-	-	-	-	-	-	682	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	202	229	-	-	-	-	-	-	-	607	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	221	232	-	-	-	-	-	-	7.96	741	-	5.2	-	-	-	-	0.14	-	-	-	40.6	-	-	-	0.9	-	281	96	117
	234	236	20	41	102	23.2	243	131	8.09	726	80	3.4	29.3	0.08	0.02	1.32	0.05	0.19	0.03	750	29.0	4.3	<0.36	<0.18	1.0	2	281	100	145

Table A.5.1. Cont'd.

Limnocorral	Sample no.	Day of year	Cl	SO_4	Na	K	Ca	Mg	pH	DIC	Spec Cond µS/cm	Susp N	TDN	NO_3	NO_2	NH_4	Susp P	TDP	SRP	DOC	Susp C	SRSi	Fe	Mn	Chl a µg/L	TSS mg/L	O_2	O_2 % Sat'n	O_2 Resp'n µmol/m³-h
HS81-6	41	171	28	196	88	21.3	235	116	-	718	95	2.1	23.6	0.07	-	0.10	0.18	0.13	-	1880	25.3	30.4	0.38	<0.36	1.0	2	-	-	-
	49	175	-	-	-	-	-	-	-	685	-	-	-	-	-	-	-	-	-	-	-	-	-	-	1.4	21	-	-	-
	57	183	17	40	106	27.3	248	108	-	822	-	5.8	18.6	0.06	0.01	0.11	0.62	0.26	0.04	600	37.0	28.9	1.65	<0.36	3.5	22	287	-	202
	73	191	20	41	103	26.1	264	110	8.25	943	91	7.4	19.3	0.03	<0.01	0.36	0.58	0.23	0.06	630	42.0	30.1	2.20	<0.36	5.1	19	285	93	166
	90	201	20	39	97	24.0	290	119	8.41	926	103	6.6	20.0	0.11	-	0.34	0.45	0.19	0.03	610	41.8	29.4	1.65	<0.36	3.7	-	278	96	353
	181	215	17	48	101	22.1	330	127	8.34	1080	102	5.9	21.4	0.02	0.01	0.23	0.48	0.32	0.06	700	38.0	27.8	3.53	<0.18	2.4	14	287	96	164
	193	226	-	-	-	-	-	-	-	1010	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	203	228	-	-	-	-	-	-	-	982	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	222	232	-	-	-	-	-	-	8.13	1092	-	5.8	-	-	-	-	0.43	-	-	-	39.8	-	-	-	1.6	-	276	94	52
	235	236	20	41	102	24.3	375	135	8.29	1120	114	4.0	21.4	0.05	0.01	0.31	0.28	0.23	0.04	720	29.6	27.7	0.44	<0.18	1.5	13	272	96	145
C81-7	42	171	39	48	87	20.8	216	108	-	688	80	2.7	22.8	0.07	-	0.38	0.21	0.16	-	690	25.8	30.4	0.38	<0.36	1.1	2	-	-	-
	50	174	-	-	-	-	-	-	-	796	-	-	-	-	-	-	-	-	-	-	-	-	-	-	1.3	3	-	-	-
	58	183	31	40	102	26.1	214	106	-	727	-	7.5	21.4	0.07	<0.01	0.15	0.23	0.23	<0.03	600	41.0	27.6	1.10	<0.36	3.5	3	295	-	222
	66	189	-	-	-	-	-	-	-	753	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	74	191	20	41	100	24.0	224	112	8.35	776	77	15.3	20.0	0.01	0.01	0.24	0.30	0.16	0.03	600	80.4	27.5	0.55	<0.36	7.1	2	300	98	418
	91	201	17	39	99	23.6	216	114	8.33	724	84	12.3	22.8	0.02	-	0.43	0.37	0.16	0.05	630	75.4	24.2	0.55	<0.36	4.9	2	296	103	588
	182	215	17	40	98	22.1	243	116	8.20	909	77	8.3	28.6	0.02	0.01	0.26	0.19	0.23	<0.03	690	56.0	16.2	<0.36	<0.18	2.2	2	296	99	346
	223	232	-	-	-	-	-	-	-	-	-	4.5	-	-	-	-	0.17	-	-	-	36.5	-	-	-	0.7	-	287	98	59
	236	236	20	41	105	23.2	235	121	8.12	687	80	4.0	31.4	0.08	0.02	1.22	0.15	0.26	<0.03	730	34.0	3.1	<0.36	<0.18	0.7	2	284	99	152
L81-8	43	171	20	49	90	21.3	219	112	-	707	81	2.3	22.1	0.07	-	0.49	0.28	0.16	-	680	24.5	30.2	0.75	<0.36	1.1	4	-	-	-
	51	175	-	-	-	-	-	-	-	712	-	-	-	-	-	-	-	-	-	-	-	-	-	-	1.5	4	-	-	-
	59	183	17	37	105	25.2	222	112	-	759	-	4.0	17.8	0.02	<0.01	0.33	0.30	0.23	0.04	560	29.3	27.8	1.10	<0.36	3.0	6	284	-	215
	67	189	-	-	-	-	-	-	-	880	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	75	191	20	40	105	26.1	230	112	7.92	780	79	5.8	20.7	0.02	0.02	0.52	0.35	0.23	0.04	570	40.1	29.3	1.10	<0.36	4.7	7	279	91	294
	92	201	20	37	99	22.8	238	117	7.94	831	90	4.9	21.4	0.02	-	0.50	0.28	0.19	0.04	600	39.0	29.5	1.10	<0.36	3.1	3	279	95	376
	183	215	17	39	99	22.1	248	125	8.07	1040	85	4.4	22.8	<0.01	0.01	0.25	0.22	0.26	<0.03	640	29.8	29.2	0.44	<0.18	1.9	-	278	94	299
	224	232	-	-	-	-	-	-	-	-	-	6.1	-	-	-	-	0.27	-	-	-	41.0	-	-	-	1.8	-	274	94	125
	237	236	20	39	105	24.9	266	129	7.93	870	89	4.9	22.8	0.01	0.01	0.54	0.21	0.29	<0.03	690	35.3	33.6	0.88	<0.18	2.2	-	271	95	260

Table A.5.2. Absorbances for 1981 limnocorrals.

Limno-corral	Sample no.	Day of year	Absorbances (10 cm path length) 455 nm			
			455 nm Centri	543 nm Unfilt	543 nm Filt	543 nm Centri
C81-1	36	171	0.055	0.100	0.038	0.021
	44	174	-	-	-	-
	52	183	0.048	0.073	0.023	0.013
	68	191	0.048	0.129	0.035	0.018
	85	201	0.041	0.102	0.025	0.013
	176	215	0.050	0.115	0.022	0.012
	217	232	-	-	-	-
	230	236	0.040	0.070	0.023	0.012
LM81-2	37	171	0.048	0.087	0.032	0.020
	45	175	-	-	-	-
	53	183	0.064	0.060	0.030	0.022
	69	191	0.076	0.154	0.044	0.031
	86	201	0.071	0.082	0.039	0.025
	177	215	0.073	0.052	0.032	0.027
	218	232	-	-	-	-
	231	236	0.068	0.069	0.031	0.023
PN81-3	38	171	0.043	0.087	0.032	0.018
	46	174	-	-	-	-
	54	183	0.046	0.072	0.023	0.015
	65	189	-	-	-	-
	70	191	0.046	0.140	0.024	0.016
	87	201	0.042	0.147	0.019	0.013
	178	215	0.040	0.171	0.018	0.010
	190	226	-	-	-	-
	200	229	-	-	-	-
	219	232	-	-	-	-
	232	236	0.040	0.150	0.027	0.013
LS81-4	39	171	0.049	0.085	0.034	0.018
	47	175	-	-	-	-
	55	183	0.050	0.113	0.040	0.016
	71	191	0.045	0.312	0.042	0.017
	88	201	0.040	0.217	0.039	0.012
	179	215	0.062	0.438	0.076	0.023
	191	226	-	-	-	-
	201	229	-	-	-	-
	220	232	-	-	-	-
	233	236	0.038	0.355	0.105	0.013
C81-5	40	171	0.044	0.083	0.037	0.018
	48	174	-	-	-	-
	56	183	0.044	0.061	0.022	0.014
	72	191	0.042	0.092	0.024	0.016
	89	201	0.037	0.085	0.018	0.010
	180	215	0.037	0.078	0.018	0.010
	192	226	-	-	-	-
	202	229	-	-	-	-
	221	232	-	-	-	-
	234	236	0.034	0.049	0.026	0.011
HS81-6	41	171	0.042	0.065	0.030	0.014
	49	175	-	-	-	-
	57	183	0.051	0.480	0.085	0.019
	73	191	0.052	0.585	0.118	0.022
	90	201	0.045	0.408	0.102	0.016
	181	215	0.069	0.042	0.109	0.028
	193	226	-	-	-	-
	203	228	-	-	-	-
	222	232	-	-	-	-
	235	236	0.032	0.452	0.162	0.010
C81-7	42	171	0.052	0.072	0.030	0.020
	50	174	-	-	-	-
	58	183	0.043	0.048	0.023	0.015
	66	189	-	-	-	-
	74	191	0.043	0.085	0.020	0.017
	91	201	0.038	0.086	0.014	0.010
	182	215	0.038	0.070	0.020	0.010
	223	232	-	-	-	-
	236	236	0.034	0.050	0.030	0.010
L81-8	43	171	0.055	0.123	0.040	0.021
	51	175	-	-	-	-
	59	183	0.048	0.081	0.032	0.018
	67	189	-	-	-	-
	75	191	0.045	0.165	0.038	0.018
	92	201	0.040	0.108	0.030	0.014
	183	215	0.039	0.088	0.036	0.012
	224	232	-	-	-	-

Table A.5.3. Chemistry data for the 1982 limnocorrals. (Units are μmoles/L unless otherwise stated).

Limnocorral	Sample no.	Day of year	Cl	SO_4	Na	K	Ca	Mg	pH	DIC	Spec Cond μS/cm	Susp N	TDN	Susp P	TDP	DOC	Susp C	Fe	Mn	Chl a μg/L	O_2	O_2 Saturation %	Absorbances (10 cm cell) 455 nm Centri	Unfilt	543 nm Filt	Centri
FMR2-1	1	181	20	39	98	24.3	310	122	7.67	-	88	13.6	22.8	0.40	0.26	1920	96.3	2.00	<0.18	1.8	-	-	0.078	0.205	0.056	0.028
	12	186	23	40	98	25.2	305	122	7.67	-	92	16.1	24.3	0.34	0.32	1070	94.6	2.00	<0.18	1.9	-	-	0.082	0.142	0.053	0.034
	32	187	-	-	-	-	-	-	7.63	-											-	-	-	-	-	-
	23	188	23	39	103	25.2	307	128	7.81	869	94	15.3	27.1	0.34	0.39	510	133.2	2.00	<0.18	2.3	257	84	0.085	0.153	0.051	0.034
	63	190	-	-	-	-	-	-	7.48	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	91	195	-	-	-	-	-	-	-	867	-	17.3	-	0.36	-	-	114.0	-	-	2.0	-	-	-	-	-	-
	116	202	23	39	104	22.9	296	128	7.59	882	94	12.1	25.7	0.48	0.29	1810	76.6	2.00	<0.18	2.4	271	89	0.075	0.089	0.038	0.028
	162	203	-	-	-	-	-	-	7.68	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	143	207	-	-	-	-	-	-	7.70	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	153	209	-	-	-	-	-	-	7.61	916	94	13.1	-	0.34	-	-	82.3	-	-	1.7	-	-	-	-	-	-
	187	215	23	39	100	25.7	312	136	7.78	830	98	19.6	25.0	0.41	0.29	700	125.6	2.50	<0.18	2.2	276	-	0.078	0.084	0.038	0.024
	266	221	-	-	-	-	-	-	7.67	864	97	18.3	-	0.48	-	-	124.6	-	-	2.4	-	-	-	-	-	-
	290	228	20	37	99	24.7	289	134	7.80	899	92	17.2	23.6	0.59	0.42	810	276.1	1.66	0.28	3.6	270	-	0.068	0.126	0.040	0.024
	325	235	17	37	107	25.8	308	140	7.87	952	97	8.4	19.3	0.39	0.45	1560	81.6	1.66	0.28	2.2	280	88	0.060	0.190	0.038	0.024
	353	240	17	37	100	24.2	291	136	7.73	883	93	15.8	25.0	0.35	0.42	600	192.8	1.10	<0.36	1.9	-	-	0.065	0.087	0.032	0.020
LM82-3	3	181	14	67	99	25.2	300	132	7.74	-	86	12.3	21.4	0.34	0.23	1170	66.9	1.50	<0.18	1.7	-	-	0.069	0.182	0.060	0.023
	13	186	25	43	97	26.1	239	94	7.29	-	75	12.1	22.8	0.31	0.29	1080	61.9	1.50	0.24	1.9	-	-	0.086	0.130	0.053	0.034
	33	187	-	-	-	-	-	-	7.02	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	25	188	23	40	99	26.6	239	94	7.42	700	75	12.1	25.0	0.28	0.29	480	63.3	1.00	0.24	2.1	259	83	0.088	0.110	0.048	0.036
	64	190	-	-	-	-	-	-	7.03	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	93	195	-	-	-	-	-	-	-	739	-	9.4	-	0.21	-	-	53.0	-	-	1.4	-	-	-	-	-	-
	118	202	23	39	97	28.0	230	98	7.36	728	75	7.4	24.3	0.19	0.42	800	47.3	1.50	<0.18	1.4	244	80	0.087	0.068	0.049	0.034
	164	203	-	-	-	-	-	-	7.30	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	145	207	-	-	-	-	-	-	7.53	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	155	209	-	-	-	-	-	-	7.58	708	76	8.3	-	0.25	-	-	50.0	-	-	1.3	-	-	-	-	-	-
	189	215	25	36	101	31.3	244	112	7.64	698	78	10.3	23.6	0.34	0.65	790	56.6	1.50	<0.18	1.7	248	80	0.084	0.057	0.034	0.028
	268	221	-	-	-	-	-	-	7.48	675	78	10.8		0.31			61.6			2.0	-	-	-	-	-	-
	292	228	20	37	99	29.8	245	117	7.72	755	76	8.6	24.3	0.57	0.71	1850	62.6	1.10	0.28	5.3	261	-	0.075	0.063	0.039	0.028
	327	235	17	36	107	32.8	250	123	7.72	765	80	12.4	24.3	0.96	0.61	1110	88.9	0.55	<0.36	6.7	263	83	0.067	0.099	0.030	0.025
	355	240	17	35	100	32.3	229	117	7.65	714	76	19.3	25.0	0.67	0.55	670	152.4	0.55	<0.36	6.7	-	-	0.067	0.067	0.028	0.022
HM82-4	4	181	11	69	97	24.3	305	124	7.63	-	88	9.7	22.1	0.30	0.26	900	53.6	1.50	<0.18	1.4	-	-	0.069	0.187	0.059	0.023
	14	186	20	40	98	25.2	171	80	6.91	-	59	11.6	22.8	0.35	0.32	650	74.3	1.50	<0.18	1.8	-	-	0.101	0.132	0.062	0.043
	34	187	-	-	-	-	-	-	6.68	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	26	188	20	40	96	26.1	164	84	7.09	536	57	11.6	25.0	0.41	0.29	740	68.6	1.50	0.24	2.2	232	75	0.095	0.119	0.054	0.038
	65	190	-	-	-	-	-	-	6.93	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	94	195	-	-	-	-	-	-	-	620	-	11.1	-	0.35	-	-	78.3	-	-	2.4	-	-	-	-	-	-
	119	202	25	36	100	33.6	153	84	6.94	573	55	14.0	27.1	0.41	0.48	660	121.2	1.50	<0.18	2.6	216	71	0.104	0.092	0.055	0.040
	165	203	-	-	-	-	-	-	6.98	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	146	207	-	-	-	-	-	-	7.21	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	156	209	-	-	-	-	-	-	7.00	572	58	21.1	-	0.74	-	-	121.9	-	-	4.8	-	-	-	-	-	-
	190	215	23	33	97	39.2	160	90	7.40	566	58	30.4	27.1	0.96	0.65	850	170.2	1.00	<0.18	7.4	205	-	0.104	0.084	0.045	0.038
	269	221	-	-	-	-	-	-	7.14	536	57	31.2	-	1.01	-	-	171.5	-	-	8.9	-	-	-	-	-	-
	293	228	20	34	106	39.4	158	86	7.35	612	56	14.9	27.8	0.98	0.65	850	95.9	0.55	0.28	7.8	215	-	0.095	0.076	0.042	0.041
	328	235	17	32	100	42.9	150	90	7.50	634	59	18.8	31.4	1.20	0.71	1080	124.7	1.10	0.28	12.6	208	66	0.098	0.148	0.049	0.043
	356	240	17	32	95	40.9	139	86	7.13	607	56	25.3	31.4	1.03	0.77	870	161.1	0.55	0.28	6.3	-	-	0.100	0.100	0.048	0.040

Table A.5.3. Cont'd.

Limnocorral	Sample no.	Day of year	Cl	SO$_4$	Na	K	Ca	Mg	pH	DIC	Spec Cond μS/cm	Susp N	TDN	Susp P	TDP	DOC	Susp C	Fe	Mn	Chl a μg/L	O$_2$	O$_2$ Saturation %	Absorbances (10 cm cell) 455 nm		543 nm	
																							Centri	Unfilt	Filt	Centri
OC82-5	5	181	20	40	102	25.2	319	130	7.72	-	88	9.7	20.7	0.31	0.23	750	65.9	1.50	<0.18	1.6	-	-	0.071	0.160	0.054	0.024
	15	186	23	42	97	25.2	307	124	7.78	-	91	10.7	20.7	0.30	0.23	1460	55.6	1.00	<0.18	1.9	-	-	0.068	0.112	0.046	0.025
	35	187	-	-	-	-	-	-	7.88	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	27	188	20	40	104	25.2	317	128	7.97	840	92	10.0	22.1	0.26	0.19	520	60.6	1.00	<0.18	1.7	287	93	0.075	0.113	0.042	0.026
	66	190	-	-	-	-	-	-	7.92	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	95	195	-	-	-	-	-	-	-	812	-	16.4	-	0.32	-	-	104.6	-	-	2.7	-	-	-	-	-	-
	120	202	23	37	102	24.7	312	130	7.93	846	94	11.6	22.1	0.27	0.19	800	66.6	0.50	<0.18	2.2	292	96	0.053	0.075	0.028	0.015
	166	203	-	-	-	-	-	-	7.84	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	147	207	-	-	-	-	-	-	8.03	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	157	209	-	-	-	-	-	-	8.03	844	93	9.0	-	0.21	-	-	58.0	-	-	1.5	-	-	-	-	-	-
	191	215	23	39	97	25.2	310	128	7.90	792	97	12.8	22.1	0.36	0.23	640	71.9	0.50	<0.18	2.1	288	92	0.049	0.048	0.018	0.010
	270	221	-	-	-	-	-	-	7.88	834	97	12.5	-	0.35	-	-	71.3	-	-	2.0	-	-	-	-	-	-
	294	228	20	37	99	24.2	289	128	7.87	880	92	6.2	21.4	0.28	0.39	610	42.0	<0.54	0.28	2.9	272	-	0.043	0.060	0.020	0.013
	329	235	17	39	102	24.2	308	134	7.85	924	94	7.1	20.7	0.36	0.36	670	53.1	0.55	<0.36	2.3	275	87	0.038	0.068	0.020	0.015
	357	240	20	37	99	25.3	289	130	7.82	892	94	12.1	23.6	0.31	0.39	590	95.8	<0.54	<0.36	2.5	-	-	0.042	0.055	0.018	0.011
C82-6	6	181	20	39	96	24.7	303	122	7.70	-	88	10.7	22.1	0.31	0.23	610	61.6	1.50	<0.18	1.4	-	-	0.071	0.173	0.058	0.025
	16	186	20	40	96	24.7	293	122	7.80	-	91	10.1	20.7	0.28	0.29	1270	55.6	1.50	<0.18	1.8	-	-	0.082	0.137	0.052	0.032
	36	187	-	-	-	-	-	-	7.80	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	28	188	23	40	97	24.7	305	126	7.92	840	92	12.0	22.1	0.28	0.23	730	64.9	1.50	<0.18	1.7	284	91	0.078	0.124	0.052	0.032
	67	190	-	-	-	-	-	-	7.73	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	96	195	-	-	-	-	-	-	-	830	-	12.3	-	0.21	-	-	74.6	-	-	1.6	-	-	-	-	-	-
	121	202	25	39	97	25.2	310	128	7.79	855	93	10.3	22.8	0.28	0.29	710	63.3	1.50	<0.18	1.9	278	91	0.068	0.090	0.044	0.022
	167	203	-	-	-	-	-	-	7.77	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	148	207	-	-	-	-	-	-	7.92	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	158	209	-	-	-	-	-	-	7.92	844	94	12.4	-	0.22	-	-	75.3	-	-	1.8	-	-	-	-	-	-
	192	215	23	40	96	24.3	305	140	7.90	783	98	14.1	22.8	0.31	0.26	770	81.3	1.00	<0.18	3.3	289	-	0.063	0.065	0.026	0.018
	271	221	-	-	-	-	-	-	7.90	834	97	19.6	-	0.40	-	-	109.2	-	-	3.9	-	-	-	-	-	-
	295	228	20	40	99	23.7	291	132	7.85	851	92	10.2	22.8	0.39	0.36	960	70.3	0.55	0.28	4.9	281	-	0.051	0.068	0.025	0.018
	330	235	17	40	110	24.2	327	150	7.92	896	96	17.1	22.8	0.59	0.39	700	114.6	0.55	<0.36	4.7	288	91	0.043	0.079	0.020	0.016
	358	240	20	40	99	24.2	286	132	7.83	866	94	31.7	24.3	1.01	0.36	640	204.0	<0.54	<0.36	3.2	-	-	0.046	0.070	0.021	0.015
HS82-7	7	181	20	39	99	24.3	312	124	7.69	-	88	12.0	21.4	0.30	0.23	1350	60.0	1.50	<0.18	1.3	-	-	0.072	0.194	0.060	0.027
	17	186	23	41	97	26.1	347	128	8.00	-	102	12.8	22.1	0.93	0.29	550	103.9	2.00	<0.18	2.1	-	-	0.087	0.660	0.121	0.035
	37	187	-	-	-	-	-	-	8.11	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	29	188	25	39	99	26.1	362	130	8.12	937	104	12.6	22.1	0.56	0.26	690	82.9	2.00	<0.18	2.0	298	96	0.077	0.600	0.109	0.030
	68	190	-	-	-	-	-	-	8.10	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	97	195	-	-	-	-	-	-	-	958	-	12.9	-	0.58	-	-	78.9	-	-	2.6	-	-	-	-	-	-
	122	202	23	39	97	26.1	415	138	8.01	1060	115	11.2	22.1	0.53	0.32	680	70.6	2.00	<0.18	3.2	276	91	0.063	0.442	0.120	0.018
	168	203	-	-	-	-	-	-	8.01	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	171	203	-	-	-	-	-	-	8.02	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	149	207	-	-	-	-	-	-	8.08	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	159	209	-	-	-	-	-	-	8.03	1080	118	14.0	-	0.61	-	-	70.3	-	-	2.9	-	-	-	-	-	-
	193	215	20	40	104	25.7	453	148	8.07	1040	123	18.2	22.1	0.81	0.32	770	115.6	1.50	<0.18	4.9	285	-	0.060	0.340	0.091	0.018
	272	221	-	-	-	-	-	-	8.02	1152	123	-	-	0.59	-	-	104.6	-	-	3.5	-	-	-	-	-	-
	296	228	<28	375	103	25.3	444	150	8.07	1176	121	8.1	20.7	0.81	0.42	3550	66.3	0.55	0.28	4.7	276	-	0.042	0.354	0.056	0.017
	331	235	20	39	106	25.3	482	165	8.20	1269	130	7.5	20.0	0.70	0.55	610	58.3	0.55	<0.36	4.1	281	89	0.040	0.389	0.074	0.015
	359	240	20	39	108	24.7	482	160	8.25	1249	129	12.5	20.0	0.74	0.45	2400	118.2	0.55	0.28	3.7	-	-	0.045	0.490	0.075	0.012

Table A.5.3. Cont'd.

Limnocorral	Sample no.	Day of year	Cl	SO_4	Na	K	Ca	Mg	pH	DIC	Spec Cond µS/cm	Susp N	TDN	Susp P	TDP	DOC	Susp C	Fe	Mn	Chl a µg/L	O_2	O_2 Saturation %	Absorbances (10 cm cell) 455 nm Centri	455 nm Unfilt	543 nm Filt	543 nm Centri
DMA2-8	8	181	17	41	97	24.7	305	120	7.76	-	88	9.6	22.1	0.26	0.29	1140	49.3	1.50	<0.18	1.6	-	-	0.070	0.146	0.055	0.025
	18	186	20	37	95	24.7	293	120	7.82	-	90	9.8	25.0	0.28	0.26	540	55.3	1.50	<0.18	1.8	-	-	0.070	0.112	0.047	0.025
	38	187	-	-	-	-	-	-	7.82	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	30	188	23	39	97	24.7	300	122	7.95	864	92	10.6	22.8	0.30	0.23	710	56.3	1.50	<0.18	1.7	291	95	0.072	0.111	0.048	0.027
	69	190	-	-	-	-	-	-	8.02	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	98	195	-	-	-	-	-	-	-	784	-	9.5	-	0.23	-	-	54.6	-	-	1.4	-	-	-	-	-	-
	123	202	20	35	97	24.3	293	122	7.80	819	90	10.4	22.1	0.25	0.29	740	64.6	2.00	<0.18	1.8	281	92	0.059	0.060	0.030	0.015
	169	203	-	-	-	-	-	-	7.78	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	150	207	-	-	-	-	-	-	7.89	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	160	209	-	-	-	-	-	-	7.99	798	93	12.1	-	0.30	-	-	73.3	-	-	1.8	-	-	-	-	-	-
	194	215	28	36	96	23.8	300	128	7.87	774	94	16.4	25.0	0.32	0.32	710	108.2	2.00	<0.18	3.6	278	89	0.054	0.058	0.020	0.014
	273	221	-	-	-	-	-	-	7.79	804	92	18.6	-	0.40	-	-	116.9	-	-	4.6	-	-	-	-	-	-
	297	228	20	37	102	23.2	283	130	7.90	832	87	8.8	23.6	0.46	0.39	1860	67.8	1.10	0.28	5.7	280	-	0.046	0.095	0.019	0.015
	332	235	17	33	99	23.7	278	125	7.95	877	91	10.3	23.6	0.56	0.55	710	103.2	1.10	0.28	5.7	280	88	0.040	0.078	0.019	0.015
	360	240	17	34	97	23.7	256	123	7.87	839	86	21.3	24.3	0.49	0.45	780	201.5	1.10	<0.36	4.5	-	-	0.045	0.065	0.020	0.015
LA2-9	9	181	20	39	100	25.2	315	122	7.81	-	88	12.6	22.1	0.28	0.26	820	64.9	1.50	<0.18	1.3	-	-	0.069	0.198	0.059	0.027
	39	187	-	-	-	-	-	-	7.80	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	31	188	23	39	101	25.7	331	124	7.83	831	93	10.8	22.8	0.28	0.26	620	55.0	2.00	<0.18	1.6	273	88	0.076	0.209	0.070	0.030
	70	190	-	-	-	-	-	-	7.88	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	99	195	-	-	-	-	-	-	-	812	-	11.4	-	0.25	-	-	57.0	-	-	2.0	-	-	-	-	-	-
	124	202	20	37	94	25.7	303	122	7.72	855	95	9.4	20.7	0.30	0.26	710	46.3	1.50	<0.18	2.4	275	89	0.066	0.160	0.059	0.024
	170	203	-	-	-	-	-	-	7.80	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	151	207	-	-	-	-	-	-	7.88	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	161	209	-	-	-	-	-	-	7.92	853	96	9.5	-	-	-	-	45.6	-	-	1.6	-	-	-	-	-	-
	195	215	20	40	99	24.7	336	134	7.90	840	98	11.7	20.0	0.35	0.23	650	66.9	1.50	<0.18	2.6	293	94	0.054	0.069	0.037	0.015
	274	221	-	-	-	-	-	-	7.78	834	97	10.0	-	0.27	-	-	52.3	-	-	2.3	-	-	-	-	-	-
	298	228	<28	354	99	24.7	321	136	7.95	899	97	4.3	20.7	0.27	0.36	2310	32.8	1.10	<0.36	1.9	273	-	0.061	0.150	0.052	0.020
	333	235	17	41	99	25.3	270	142	7.93	952	100	4.7	20.0	0.31	0.45	1140	34.6	1.10	<0.36	2.0	267	84	0.075	0.161	0.055	0.030
	361	240	17	42	95	23.2	319	138	7.90	901	100	9.4	20.7	0.28	0.39	750	74.5	1.10	<0.36	2.1	-	-	0.080	0.150	0.052	0.026

Table A.5.4. Chemistry data for the 1983 limnocorrals. (Units are µmoles/L unless otherwise stated).

Limnocorral	Sample no.	Day of year	Cl	SO_4	Na	K	Ca	Mg	pH	Alk µeq/L	DIC	Spec Cond µS/cm	Susp N	TDN	Susp P	TDP	DOC	Susp C	SRSi	Fe	Mn	Chl a µg/L	TSS mg/L	O_2	Absorbances (10 cm path length)			
																									455 nm		543 nm	
																									Centri	Unfilt	Filt	Centri
DFM83-1	429	182	-	-	-	-	-	-	7.20	-	867	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	443	188	25	45	103	25.8	356	152	7.55	830	-	89	4.2	35.7	0.32	0.45	820	31.6	25.4	3.35	0.26	2.6	4	270	0.088	0.110	0.040	0.039
	473	202	23	45	103	26.4	266	116	7.57	830	834	88	33.3	44.3	0.88	0.32	840	386.2	16.9	3.19	<0.18	4.8	10	267	0.078	0.164	0.034	0.025
	489	213	-	-	-	-	-	-	-	-	723	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	541	217	20	45	101	28.0	271	119	7.25	870	984	88	7.9	47.1	0.44	0.61	950	54.1	14.8	5.55	0.54	4.0	5	203	0.100	0.144	0.059	0.038
	556	224	-	-	-	-	-	-	7.65	-	965	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	586	236	-	-	-	-	-	-	-	-	830	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	595	237	25	40	106	28.0	304	120	-	825	792	84	14.0	49.3	0.65	0.65	830	153.4	30.4	12.49	0.27	-	6	297	0.148	0.212	0.085	0.050
SP83-2	430	182	-	-	-	-	-	-	7.30	-	849	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	444	188	25	47	100	29.9	367	160	7.10	835	-	92	5.4	33.5	0.44	0.61	930	43.1	19.8	2.79	1.05	1.2	2	177	0.142	0.158	0.094	0.087
	474	202	20	42	102	36.2	282	122	6.86	885	1018	93	29.4	44.3	1.25	0.81	1870	118.2	21.0	14.65	2.35	2.4	6	22	0.232	0.324	0.170	0.150
	490	213	-	-	-	-	-	-	-	-	893	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	542	217	23	41	102	54.3	291	126	6.60	990	1144	96	35.0	57.1	3.63	1.03	2330	274.8	26.6	22.88	3.77	31.6	8	<10	0.349	0.504	0.250	0.238
	557	224	-	-	-	-	-	-	6.95	-	1220	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	587	236	-	-	-	-	-	-	-	-	1145	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	596	237	31	25	103	60.6	282	126	-	930	1078	96	37.1	52.8	2.70	0.77	1390	341.7	35.6	9.71	4.57	-	9	33	0.323	0.580	0.224	0.199
LM83-3	431	182	-	-	-	-	-	-	7.00	-	748	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	436	187	-	-	-	-	-	-	7.62	-	660	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	445	188	25	46	102	24.3	289	145	7.25	640	-	70	2.7	28.6	0.22	0.39	790	22.3	28.8	1.12	0.26	0.8	2	259	0.093	0.086	0.046	0.043
	475	202	25	45	99	28.0	200	95	7.44	570	646	73	3.6	30.7	0.34	0.39	640	27.5	31.4	1.91	<0.18	1.1	2	240	0.093	0.096	0.053	0.037
	491	213	-	-	-	-	-	-	-	-	567	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	543	217	23	43	101	32.6	215	104	7.25	705	738	75	3.4	67.8	0.35	0.55	910	28.1	33.4	0.69	0.27	1.8	2	220	0.092	0.092	0.050	0.038
	558	224	-	-	-	-	-	-	7.30	-	720	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	588	236	-	-	-	-	-	-	-	-	699	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	597	237	25	39	103	32.6	209	104	-	735	675	72	5.2	38.5	0.49	0.45	830	42.0	29.2	0.69	0.27	-	2	269	0.079	0.069	0.034	0.025
C83-4	437	187	-	-	-	-	-	-	7.83	-	771	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	446	188	25	45	102	23.8	345	152	7.60	845	-	88	7.1	57.1	0.44	0.32	740	49.3	17.7	0.56	0.26	2.8	3	286	0.078	0.082	0.029	0.028
	476	202	23	45	98	23.3	271	116	7.68	865	779	91	6.3	30.7	0.44	0.36	780	52.8	8.4	5.73	<0.18	3.2	2	269	0.062	0.078	0.022	0.019
	492	213	-	-	-	-	-	-	-	-	694	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	544	217	23	44	100	25.4	278	116	7.54	875	852	89	4.7	60.7	0.35	0.32	820	39.1	7.1	<0.72	<0.18	2.0	1	249	0.058	0.062	0.024	0.022
	561	226	-	-	-	-	-	-	8.30	-	834	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	589	236	-	-	-	-	-	-	-	-	817	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	598	237	25	40	102	25.4	275	117	-	850	816	85	4.3	42.8	0.39	0.45	700	41.8	17.6	1.39	0.54	-	2	266	0.058	0.085	0.029	0.015
OSA83-5	432	182	-	-	-	-	-	-	7.70	-	830	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	438	187	-	-	-	-	-	-	7.91	-	779	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	447	188	25	46	100	23.8	345	145	7.70	845	-	92	3.6	52.1	0.27	0.36	710	26.5	15.9	<0.54	0.26	1.0	3	295	0.069	0.087	0.027	0.027
	477	202	23	46	100	23.3	295	120	7.81	890	817	95	5.1	32.1	0.30	0.32	740	34.8	6.9	0.64	<0.18	2.1	1	271	0.057	0.072	0.022	0.018
	493	213	-	-	-	-	-	-	-	-	723	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	545	217	23	43	98	25.9	311	130	7.62	945	934	98	5.4	100.6	0.25	0.45	540	35.8	5.0	0.69	0.27	2.4	2	249	0.052	0.072	0.031	0.019
	562	226	-	-	-	-	-	-	7.95	-	956	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	590	236	-	-	-	-	-	-	-	-	996	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	599	237	25	40	100	25.9	311	132	-	1005	928	99	5.7	39.3	0.39	0.52	690	36.3	11.5	3.47	0.54	-	2	271	0.058	0.093	0.042	0.015

Table A.5.4. Cont'd.

Limnocorral	Sample no.	Day of year	Cl	SO4	Na	K	Ca	Mg	pH	Alk µeq/L	DIC	Spec Cond µS/cm	Susp N	TDN	Susp P	TDP	DOC	Susp C	SRSi	Fe	Mn	Chl a µg/L	TSS mg/L	O_2	Absorbances (10 cm path length) 455 nm Centri	Unfilt	543 nm Filt	Centri
OM83-6	433	182	-	-	-	-	-	-	7.50	-	794	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	448	188	25	45	100	23.8	345	160	7.58	845	-	89	3.9	47.8	0.27	0.36	760	26.1	27.2	0.56	<0.18	1.1	2	275	0.075	0.078	0.030	0.023
	478	202	23	44	101	25.4	262	114	7.62	835	791	90	4.3	30.7	0.25	0.32	770	42.0	27.3	1.91	<0.18	3.4	1	270	0.068	0.059	0.030	0.030
	494	213	-	-	-	-	-	-	-	-	680	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	546	217	23	47	99	25.9	291	123	7.62	780	861	89	2.9	60.0	0.18	0.32	770	26.8	24.9	1.39	0.27	2.0	2	253	0.060	0.050	0.026	0.022
	563	226	-	-	-	-	-	-	7.80	-	834	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	591	236	-	-	-	-	-	-	-	-	791	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	600	237	25	40	103	25.9	266	123	-	865	802	85	1.5	35.0	0.14	0.36	660	18.3	21.8	2.08	0.27	-	1	269	0.051	0.035	0.021	0.013
LM83-7	434	182	-	-	-	-	-	-	7.05	-	803	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	439	187	-	-	-	-	-	-	7.53	-	660	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	449	188	23	46	100	24.3	289	138	7.32	660	-	71	3.7	71.4	0.23	0.39	770	24.0	28.9	0.56	0.26	0.8	2	261	0.089	0.094	0.046	0.031
	479	202	23	45	104	26.4	229	105	7.40	680	667	75	2.5	30.0	0.21	0.45	1250	20.8	32.0	1.91	<0.18	0.9	2	246	0.087	0.096	0.054	0.032
	483	203	-	-	-	-	-	-	-	-	685	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	550	216	-	-	-	-	-	-	-	-	729	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	547	217	23	43	99	31.6	242	110	7.35	750	765	78	5.2	35.0	0.40	0.23	1090	46.5	33.0	1.39	0.27	2.0	3	229	0.077	0.088	0.045	0.030
	564	226	-	-	-	-	-	-	7.30	-	746	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	592	236	-	-	-	-	-	-	-	-	725	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	601	237	25	40	101	33.6	247	117	-	775	736	78	6.4	41.4	0.65	0.58	820	51.8	17.6	0.69	0.27	-	3	271	0.070	0.108	0.037	0.019
LMS83-8	440	187	-	-	-	-	-	-	7.58	-	634	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	450	188	23	46	101	25.8	289	142	7.32	660	-	71	4.9	75.7	0.45	0.45	780	30.3	29.2	1.12	0.26	1.1	8	262	0.093	0.272	0.064	0.033
	480	202	23	45	100	27.4	235	107	7.39	655	659	73	3.1	34.3	0.41	0.48	840	37.6	34.3	2.55	<0.18	1.7	6	235	0.094	0.182	0.065	0.034
	484	203	-	-	-	-	-	-	-	-	671	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	551	216	-	-	-	-	-	-	-	-	688	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	548	217	23	43	103	31.6	229	111	7.48	720	715	77	9.3	83.5	0.66	0.48	900	58.6	35.3	1.39	0.27	9.3	4	245	0.080	0.178	0.049	0.032
	565	226	-	-	-	-	-	-	7.70	-	720	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	593	236	-	-	-	-	-	-	-	-	708	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	602	237	25	40	101	33.6	235	113	-	750	698	76	6.0	41.4	0.67	0.55	780	50.5	20.9	1.39	<0.18	-	8	271	0.071	0.245	0.049	0.019
L83-9	441	187	-	-	-	-	-	-	7.80	-	779	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	451	188	25	44	100	23.8	390	163	7.65	860	-	91	3.2	28.6	0.25	0.36	680	20.5	25.2	1.12	0.26	2.9	5	281	0.072	0.162	0.051	0.023
	481	202	23	43	102	24.8	302	122	7.72	865	796	92	3.6	31.4	0.21	0.32	730	23.6	26.8	1.91	<0.18	1.7	2	272	0.061	0.104	0.043	0.020
	485	203	-	-	-	-	-	-	-	-	796	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	552	216	-	-	-	-	-	-	-	-	802	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	549	217	23	43	94	25.4	280	117	7.70	865	825	89	2.9	30.7	0.21	0.39	680	20.8	28.9	0.69	<0.18	2.2	3	258	0.058	0.108	0.048	0.023
	566	226	-	-	-	-	-	-	7.80	-	790	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	594	236	-	-	-	-	-	-	-	-	822	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	603	237	25	42	95	25.4	289	123	-	905	830	89	2.6	37.8	0.21	0.42	630	20.6	33.9	1.39	<0.18	-	3	270	0.074	0.120	0.046	0.012

Table A.5.5. *In situ* respiration in the limnocorrals through the night of Aug 19-20, 1981.

Limnocorral	Sampling time		Time Between Samples	Depth	Oxygen Conc (μmol/L)		Respiration Rate
	Aug 19	Aug 20	(hr:min)	(m)	Aug 19	Aug 20	(μmol/m^3·h)
C81-1	21:45	5:15	7:30	0	291	279	1601
				1.5	291	279	1540
				2.5	288	274	1903
LM81-2	22:05	5:30	7:25	0	270	240	4183
				1.5	262	232	4118
				2.5	231	215	2288
PN81-3	22:20	5:40	7:20	0	331	312	2721
				1.5	331	311	2874
				2.5	322	311	1615
LS81-4	22:40	5:55	7:15	0	270	265	704
				1.5	270	264	788
				2.5	262	262	-78
C81-5	22:50	6:05	7:15	0	287	277	1376
				1.5	282	275	941
				2.5	279	274	652
HS81-6	23:05	6:20	7:15	0	277	270	1037
				1.5	263	270	-945
				2.5	268	267	54
C81-7	23:20	6:30	7:10	0	290	283	1039
				1.5	287	283	575
				2.5	282	283	-119
L81-8	23:40	6:50	7:10	0	277	274	379
				1.5	275	267	1097
				2.5	264	262	240

METHYLATION, DEMETHYLATION, AND BIO-ACCUMULATION OF MERCURY IN LAKES AND RESERVOIRS OF NORTHERN MANITOBA, WITH PARTICULAR REFERENCE TO EFFECTS OF ENVIRONMENTAL CHANGES CAUSED BY THE CHURCHILL-NELSON RIVER DIVERSION

Final report on Project number 2.3 of the study of
mercury in northern Manitoba carried out in
accordance with the Canada-Manitoba Agreement

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SPECIAL NOTE

This report was prepared under the terms of the Canada-Manitoba Agreement on the Study and Monitoring of Mercury in the Churchill River Diversion. Any conclusions, recommendations, or opinions expressed herein are those of the author and are not necessarily the same as those of the Steering Committee for this Agreement.

"We were contented to let things remain as the Great Spirit made them. [The white men] were not, and would change the rivers if they did not suit them."

Hin-mah-too-yah-lat-kekht ("Chief Joseph"),
Nez Perce Tribe of North American Indians,
Oregon

SUMMARY OF CONCLUSIONS AND RECOMMENDATIONS

I. Conclusions.

(1) The creation of hydroelectric reservoirs in northern Manitoba by the impoundment and diversion of Churchill River water, enlargement of riverine lakes, and flooding of adjacent land has resulted in elevated net rates of methyl mercury production by bottom-dwelling microbes in those reservoirs. The effect is most pronounced in regions of highly organic new (post-impoundment) lake bottom directly overlying submerged land, though it is detectable in regions of older (pre-impoundment) lake bottom too, and it tends to be greatest in reservoirs where the rise in lake level and areal extent of flooding have been greatest. This phenomenon is probably the principal cause of the sharp rise in the mercury content of fish muscle that occurred in the reservoirs soon after the diversion took place.

(2) The observed enhancement of microbial methyl mercury production in lake environments affected by flooding is due primarily to stimulation of heterotrophic microbial activity by the abundant organic matter - notably remains of plants - in the zone of submerged land. The increased rate of mercury methylation resulted principally from two related factors: (a) utilization of organic substances as nutrient substrates by methylating microbes, and (b) creation of anaerobic or oxygen-poor conditions, which favour mercury methylation, owing to decomposition of the organic matter. An increase in the supply of unmethylated mercury (an essential precursor of methyl mercury) owing to incorporation of decomposing mercury-bearing terrestrial organic matter into the aquatic environment has been a contributing factor, but it is probably of secondary importance.

(3) The buffering action of carbonate minerals in the glacial clay deposits of northern Manitoba fosters methylation (and demethylation) by

keeping ambient pH values close to 7. According to experimental results, the clay minerals themselves have little or no effect on rates of methylation but may greatly accelerate demethylation. In Southern Indian Lake, detrital clay from the Churchill River and from shoreline deposits probably has the net effect of depressing methyl mercury production somewhat - possibly by dilution of organic nutrients, and perhaps partly by stimulation of demethylating microbes. Effects of clay may help to explain why Southern Indian Lake sediments have lower rates of methyl mercury production than Notigi Reservoir sediments, although the principal reason for the difference is judged to be the greater proportion of recently submerged land and terrestrial organic matter in Notigi Reservoir.

(4) In general, methylation of mercury by microbes in the sediments is optimized by (a) an ample supply of labile organic matter such as decomposing remains of terrestrial plants or lacustrine algae, (b) absence or paucity of oxygen, (c) absence or paucity of sulfides such as H_2S or FeS, and (d) an ambient pH in the range 6.0-7.5. Consequently, the flooded land of reservoirs in northern Manitoba provides ideal environmental conditions for methylation. Nevertheless, methylation occurs within a wide range of trophic conditions and Eh and pH levels, and is totally suppressed only under extreme conditions - for instance, under an atmosphere of H_2S or at a pH $\leq$ 5. This apparent tolerance of environmental variability reflects the fact that methylation is carried out by different kinds of microbes which differ from one another in their ecological requirements; for instance, the methylating microbes evidently include aerobic as well as anaerobic (or facultatively anaerobic) species, and the anaerobic methylators probably include methanogenic bacteria as well as microbes that do not generate methane.

(5) Methylation is kept in check by demethylation, which prevents unlimited, uncontrolled build-up of methyl mercury concentrations, even under ideal conditions for methylation. With increasing abundance of organic nutrients in bottom sediment, or with increasing time after introduction of a given quantity of organic substrate into the sediment, the methyl mercury content of the sediment typically rises, peaks, and then declines.

(6) Like methylation, demethylation is catalyzed by a variety of microbes, including aerobic as well as anaerobic (or facultatively anaerobic) species, and methanogenic bacteria as well as anaerobic species that do not generate methane, and it occurs within a wide range of trophic conditions and Eh and pH values. Optimal conditions for demethylation include an ambient pH in the range 6.0-7.5 and anaerobic conditions without nutrient enrichment, although aerobic and anaerobic conditions with nutrient enrichment are also highly favourable. Aerobic conditions, a pH in the range 6-9, and the presence of organic substrates should foster demethylation at the expense of methylation.

(7) The methyl mercury levels in bottom sediments and water, and the mercury content of walleye muscle, in the Churchill-Rat River system appear to depend primarily on two factors: (a) the abundance of "available" or easily solubilized, exchangeable forms of inorganic mercury in the sediments, and (b) the rate at which sedimentary microbes are able to convert these available inorganic mercury species into methyl mercury. Inundation of land due to hydroelectric development has increased the methyl mercury levels in the lakes, and therefore the mercury levels in the fish, not only by stimulating the activities of methylating microbes in the sediments (principally owing to nutrient enrichment) but also by increasing

the pool of readily mobilized, and therefore readily methylated, inorganic mercury species in those sediments (probably owing to release of mercury from decomposing submerged terrestrial organic matter), the observed effects tending to be most pronounced in regions where the extent of flooding has been greatest. The availability of inorganic mercury is probably enhanced by aeration, and hence is greatest in the "main-stem" zone of the river-lake system, where current action is particularly vigorous. Sulfide and selenium, which are most abundant in the less rapidly flushed "backwater" regions, appear to depress the availability of mercury for methylation and biological uptake.

(8) It seems doubtful whether a programme of remedial action for dealing with the mercury problem is feasible or even desirable. The fact that mercury can be methylated by a number of different types of microbe under a wide range of environmental conditions appears to rule out the possibility of finding a practical method for suppressing methylation entirely. Drastic measures would be required to achieve this effect, and even if such measures were technically and economically feasible, they would not be advisable because they would undoubtedly have extremely undesirable ecological consequences. At best, deliberate modification of aquatic or sedimentary environments might accomplish a measurable reduction in the net rate of methyl mercury production. Adjustment of river flow to promote mixing and aeration (assuming that such a measure were practicable in the first place) could conceivably ameliorate the mercury problem by lowering methylation rates and promoting demethylation, besides hastening the decomposition of the labile organic matter that nourishes the methylating microbes, although this effect might well be offset to a greater or lesser extent by enhanced availability of inorganic mercury

resulting from aeration. Increased loading of clay and silt might also be beneficial. However, it would be necessary to perform further studies taking into account not only the question of technical feasibility and probable degree of success but also the projected cost, the natural rate of recovery of the river-lake system under existing conditions, and possible negative side-effects of remedial action. Cost alone could well be a prohibitive limiting factor, considering the huge size of the river-lake system, and if the natural rate of recovery is sufficiently rapid, such expensive and Herculean efforts will not be needed anyway. Moreover, unexpected harmful side-effects often result from environmental perturbations caused by misguided human interference with Nature (the mercury problem itself being a typical example). For all of these reasons, the path of remedial action has questionable merit and should be pursued with caution, if at all. Perhaps the wisest approach would be to leave the system alone and let it recover on its own. Above all, it is hoped that the results of this study will serve as a warning that will help to prevent future environmental damage of the sort that has been caused by the exploitation of hydroelectric power in northern Manitoba.

II. Recommendations.

(1) Since it is not known how much longer the mercury problem will persist or precisely how the future course of events in the reservoirs will unfold, physicochemical and biological monitoring and research should be done on a long-term basis to observe the further post-impoundment evolution of the river-lake system; this should be continued until the system recovers or attains a new steady state. The information obtained would be useful from the standpoint of predicting recovery rates and evolutionary stages of new reservoirs. Such studies should include estimation of

mercury levels in fish, other organisms, and sediments, assays of the methyl mercury producing capability of sediments, and determination of organic matter, nitrogen, and oxidation-reduction conditions in water and surface sediments. Examination of the health, mercury body burdens, and social and economic condition of human populations affected by the Churchill-Nelson River diversion should be part of any such programme. Future studies should not be limited to mercury, as the mercury problem is only one aspect of the ecological damage and human distress caused by impoundment. Any future interdisciplinary study programme should embrace the full spectrum of ecological effects of reservoir creation - including effects on human ecology - and seek to integrate the diverse elements of this complex environmental problem.

(2) No measures for amelioration or restoration of the river-lake system are recommended at this time, as it is doubtful whether such measures would be practical, useful, or even ecologically desirable. The most sensible approach would probably be to leave the river-lake system alone and let it stabilize through natural processes without further human intervention. On the other hand, small-scale pilot projects, including field experiments, to investigate possible remedial methods, such as (1) enhancement of mixing and aeration by manipulation of river flow and (2) increased loading of clay and silt, may be worth considering.

(3) The lessons learned from the Churchill-Nelson diversion are universally, or at least widely, applicable and should be taken into account whenever and wherever reservoir creation is contemplated in the future. Phenomena such as stimulation of methyl mercury production by organic matter are basic and universal; and, not surprisingly, a survey of the literature on reservoirs strongly suggests that the general conclusions

of the northern Manitoba mercury study regarding effects of impoundment on the biogeochemistry of mercury apply to all or most newly created reservoirs, regardless of their geographical or environmental setting. Knowing the probable consequences of impoundment, responsible environmental stewardship will require that decision makers weigh very carefully the advantages and disadvantages of proceeding with reservoir creation in the case of any river system where fishing is important or essential to the livelihood of local residents. If the decision is made to proceed with impoundment in spite of the likelihood that fishing grounds will be adversely affected as a result, then removal of shoreline vegetation and soil prior to impoundment and flooding may be a worthwhile precautionary measure (Dickson, 1972) if it is feasible, and if the advantage gained thereby is not offset by objectionable ecological side-effects.

INTRODUCTION

In the mid-1970's Manitoba Hydro, with the approval of the Manitoba Government, carried out the ambitious and controversial hydroelectric project known as the Churchill-Nelson River diversion, whereby the waters of the Churchill River were impounded and diverted southward from Southern Indian Lake into the Rat-Burntwood River system by way of an artificial channel (Fig. 1) and thence into the Nelson River. As a result, a number of riverine lakes, including Southern Indian Lake and the lakes situated along the Rat River, have been enlarged to form reservoirs, and the surrounding land has been more or less extensively flooded. The decision to proceed with hydroelectric development was made in spite of warnings about probable ill-effects on the fisheries of the region (Dickson, 1972).

The widespread inundation of the densely forested land bordering the rivers and lakes has had a marked impact on the physicochemical properties and biota of the lakes, and the pre-existing ecological balance of the region has been upset (Hecky and Harper, 1974; Hecky _et al._, 1979; Guilbault _et al._, 1979; Bodaly _et al._, 1980; Jackson and Hecky, 1980; Canadian Journal of Fisheries and Aquatic Sciences _41_: 548-732 (1984); Jackson, 1984b). Deleterious environmental changes such as increased turbidity have caused a drastic decline in the quantity and quality of the commercial whitefish catch harvested from Southern Indian Lake (Bodaly, 1982; Bodaly _et al._, 1984b), and this has brought economic and social stress to a local band of Cree Indians who had been accustomed to supporting themselves by fishing.

A major effect of the Churchill-Nelson River diversion, and the one that is of principal concern in this report, was a rapid and substantial rise in the mercury (Hg) content of muscle tissue in several species of fish (walleye, northern pike, and lake whitefish) in the affected lakes (Bodaly and Hecky, 1979; McGregor, 1980; Bodaly et al., 1984a). This is cause for serious concern, as the mean Hg concentrations in pike and walleye have risen above the 0.5 μg/g legal limit for fish marketed for human consumption in Canada, and they show no sign of declining (although Hg levels in whitefish in Southern Indian Lake started decreasing progressively with time soon after the initial upsurge and have almost returned to baseline levels (Bodaly et al., 1984a)). Besides being injurious to the regional economy, the Hg poses a threat to the health of people who habitually consume locally caught fish.

The available information on the lakes and reservoirs of northern Manitoba proves beyond reasonable doubt that the Hg problem has resulted from environmental alteration caused by the Churchill-Nelson River diversion. Hg data for pike, walleye, and whitefish populations of Southern Indian Lake before and after the diversion (which occurred in 1976) indicate a sharp rise in mean Hg content of muscle tissue within about 2 years following diversion (McGregor, 1980; Bodaly et al., 1984a). Post-diversion Hg levels in fish inhabiting lakes enlarged by inundation of adjacent land are much higher than Hg levels in Granville Lake, a Churchill River lake which is situated above the inflow to Southern Indian Lake and has not been affected by the diversion, and the Hg levels in reservoir fish tend to be highest where flooding has been most extensive (McGregor, 1980; Bodaly et al., 1984a; Jackson, Ms. in preparation). Besides, there are no known point-sources of Hg contamination in the region; atmospheric pollution probably accounts for a modest, gradual accumulation of Hg observed in pike and walleye prior to diversion (Bodaly et al., 1984a), but this cannot account for the marked and abrupt increase that occurred soon after impoundment. The available evidence supports the conclusion that the observed increase in rates of Hg bio-accumulation has resulted primarily from stimulation of the growth and activity of Hg-methylating micro-organisms by nutrients - notably organic matter - incorporated into the aquatic environment from recently submerged soil and vegetation (Bodaly _et al._, 1984a; Jackson, Ms. in preparation), and release of nonmethyl Hg from diffuse natural sources (notably organic accumulations) in flooded land is probably a contributing factor; however, local variations in physicochemical environment as well as apparent differences in the Hg-cycling mechanisms of different types of aquatic organism complicate the biogeochemical picture considerably (Jackson, Ms. in preparation). But whatever the causes of the observed effects, the Hg problem in northern Manitoba is neither surprising nor unique, for there has been a mounting weight of published evidence that reservoir formation is generally followed by an increase in the rate of Hg accumulation by fish inhabiting the impounded waters (Gebhards _et al._, 1971; Potter _et al._, 1975; Abernathy and Cumbie, 1977; Bruce and Spencer, 1979; Cox _et al._, 1979; Meister _et al._, 1979; Bodaly and Hecky, 1980; McGregor, 1980; Waite _et al._, 1980; Bodaly _et al._, 1984a; Jackson, 1984b and Ms. in preparation). Because this phenomenon has occurred repeatedly in newly formed reservoirs located in widely separated, environmentally diverse regions where no anthropogenic or natural point-sources of Hg contamination are known to exist, and has not been observed in nearby water bodies unaffected by impoundment and flooding, it is believed to be a universal (or at least a widespread) side-effect of reservoir formation - a result of (1) environmental changes leading to enhancement of the mobility and bioavailability of whatever Hg happens to be present in the aquatic environment (for instance, by stimulation of microbial methylation of Hg owing to an influx of nutrients from flooded land), or (2) an increase in the total abundance of Hg owing to introduction of Hg into the aquatic environment from diffuse natural sources in flooded land, or (3) both. Until the recent investigation of Hg in the reservoirs of northern Manitoba, however, detailed knowledge of the processes responsible for the enhanced bio-accumulation of Hg in new reservoirs has apparently been lacking.

In 1983 the governments of Canada and Manitoba signed an agreement to undertake a joint investigation of the Hg problem in northern Manitoba. The study was divided into several independent but interrelated research projects to be conducted by different research groups under the administration of a steering committee. This report presents the principal results of one of these projects, together with conclusions and recommendations. The purpose of the project was to establish the causes of the Hg problem, to provide rigorous, detailed data on the processes involved, and to make recommendations pertaining to future environmental management decisions. Attention was focused on the influence of selected physicochemical variables on rates of methyl mercury (CH_3Hg^+) synthesis and decomposition (Hg methylation and demethylation, respectively) by microorganisms in lake and reservoir sediments of the Churchill-Rat River system, with particular emphasis on effects of environmental changes due to impoundment and flooding. In general, the lipophilic species CH_3Hg^+ is of key importance in the biogeochemical cycle of Hg in aquatic ecosystems because of its high degree of bio-availability and toxicity and its preferential accumulation in the tissues of fish (Jernelöv, 1972).* Synthesis of CH_3Hg^+ by methylation of inorganic bivalent Hg is performed by a variety of free-living microbes - mainly in surficial bottom sediments - under a wide range of environmental conditions, and similarly, many different kinds of microbe are capable of demethylating CH_3Hg^+ (Fagerström and Jernelöv, 1972; Jernelöv, 1972; Shariat et al., 1979). CH_3Hg^+ levels in sediments and water reflect a balance between methylation and demethylation processes.

The basic approach adopted in this study was to combine analysis of field samples with a series of laboratory experiments designed to determine (1) effects of organic substrates, oxidation-reduction conditions, pH, clay-sized minerals, and different concentrations of inorganic bivalent Hg on rates of Hg methylation and demethylation by microbes in sediments, and (2) differences in rates of methylation in sediments from different sampling sites incubated under uniform, defined conditions (cf. Hecky et al., 1986, and Ramsey and Ramlal, 1986). samples were collected from "reference" lakes unaffected by impoundment and from reservoir sites representing a range of environmental conditions and varying degrees of environmental impact due to impoundment. This report presents highlights of the study, but no attempt is made to cover the large volume of observational and experimental data exhaustively; details not included here can be furnished on request. The report deals with results of research conducted during the period August, 1983 - March, 1985, together with selected data from a closely related study performed during 1981 - 1983 (prior to ratification of the Canada-Manitoba agreement).* The earlier study is treated comprehensively in a separate paper to be submitted shortly for publication (Jackson, Ms. in preparation). When the paper is in its final form, preprints or reprints will be made available to the steering committee and to other scientists in the study group. Meanwhile, suffice it to say that the results and conclusions given in that paper are in essential agreement with those given in the present report. In both the present report and the manuscript in preparation, the writer has attempted to achieve a synthesis of his own biogeochemical research results with other workers' data on Hg levels in the tissues of fish inhabiting the lakes of the study area (Bodaly et al., 1984a; McGregor, G.W.G., unpublished data, personal communication).

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*The lipophilic character of CH_3Hg^+ is conducive to efficient bioaccumulation and a long 1/2-life in fish tissues. There is indirect evidence, however, suggesting that the CH_3Hg^+ in the whitefish of Southern Indian Lake, unlike the CH_3Hg^+ in walleye and pike, is largely converted to readily excreted hydrophilic complexes (Jackson, Ms. in preparation; also, see below).

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*Some of the data from this previous study have also been utilized by Warwick (1986), who initially participated in the project.

MATERIALS, METHODS, AND DESCRIPTION OF THE FIELD AREA

Field area and sampling sites.

The field area (Fig. 1) is located between latitudes 55° and 57° North in a region of low relief, moist sub-Arctic continental climate, podzolic soil, and Boreal forest dominated by black spruce (*Picea mariana*). The bedrock consists of Precambrian igneous and metamorphic rocks largely overlain by Pleistocene glacial sediments, including a considerable amount of varved glacio-lacustrine silty clay. Owing to the buffering action of carbonates in the Pleistocene deposits, surface waters of the region commonly have neutral to weakly alkaline pH values (Jackson and Hecky, 1980) despite an abundance of humic matter derived from local soil and muskeg.

During August 12-15, 1983, samples of sediment and water were collected from twelve lake and reservoir sites in the Churchill-Rat River system (Fig. 1). The sampling sites, and the changes in water level and surface area which occurred there as a result of impoundment (Bodaly *et al.*, 1984a), were as follows:

I. "Reference," or "control," lakes unaffected by hydroelectric development (no change in water level and surface area):

- Granville Lake
- East Mynarski Lake

II. Reservoirs:

1. Southern Indian Lake (3 m rise in water level; 21% increase in mean surface area).
    - South Bay

- Long Bay* (central basin)
- Long Bay* (shallow cove)
- "Methyl Bay" (zone of flooded forest)
- "Methyl Bay" (offshore, "normal" lacustrine zone beyond the zone of flooded forest).

2. Lakes of the Rat River system and Mynarski chain, collectively called Notigi Reservoir (6.9-15.2 m rise in water level, depending on the individual lake; 282% increase in mean surface area).
    - Notigi Lake (east basin)
    - Notigi Lake (west basin)
    - Rat Lake
    - West Mynarski Lake
    - Central Mynarski Lake.

On June 5 and 6, 1984, additional samples were collected from Granville Lake, East Mynarski Lake, and both Methyl Bay localities.

Granville Lake was designated a "control" region representing pre-impoundment baseline conditions for purposes of comparison with Southern Indian Lake and the Notigi Reservoir lakes. East Mynarski Lake is in a class by itself, being distinguished by anomalously high levels of biological productivity owing to favourable local conditions (Hecky, R.E., personal communication). Therefore, East Mynarski Lake was not suitable for utilization as a control (except with respect to the other Mynarski lakes); but, because of its abundant microflora, sediment from that lake was used most frequently for methylation/demethylation experiments (see below). The flooded forest site in Methyl Bay was the only reservoir site located directly over a recently flooded land surface; all other sites were in offshore, and hence more typically lacustrine, environments. Consequently, comparison of data for the "flooded forest" and "offshore" Methyl Bay sites and the Granville Lake site is especially instructive for assessment of effects of flooding. It is also useful to compare the offshore sites of Notigi Reservoir, Southern Indian Lake, and Granville Lake, as they represent widely differing degrees of environmental impact due to flooding, the rise in water level ranging from 0 to 15.2 m, and the severity of impact increasing in the order Granville Lake < Southern Indian Lake < Notigi Reservoir.

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*Also called Wupaw Bay.

Owing to local environmental peculiarities, however, regional variations in the physicochemical and biological properties of the lakes and reservoirs are not necessarily expressible as simple functions of the extent of flooding (Jackson, Ms. in preparation). Among these complicating circumstances are the following: (1) Some sampling sites were in well-flushed "main-stem" regions of the river system, while others were in less rapidly flushed "backwater" regions where the influence of local drainage is more pronounced (cf. Jackson and Hecky, 1980). (2) Unlike the other sampling regions, South Bay is bordered by Pleistocene silty clay deposits which are being eroded into the water, causing greatly elevated turbidity. (This process was intensified when the lake level was raised but existed previously). (3) Most of the suspended load of the Churchill River settles out above the inflow to Notigi Reservoir; therefore, Granville Lake and Southern Indian Lake are richer in fluvial mineral detritus than Notigi Reservoir, whereas Notigi Reservoir (especially the backwater zone) is richer in organic substances, including allochthonous humic matter. (4)

Granville Lake and the west basin of Notigi Lake have been found to have certain anomalies in common, such as abundant manganese-iron nodules, which set them apart from the other sampling regions (Jackson, Ms. in preparation). (5) East Mynarski Lake, as mentioned above, is exceptionally productive. West Mynarski Lake is also very productive, though somewhat less so.

Collection and analysis of field samples.

At each sampling site water and sediment samples were collected for analysis in order to characterize the environmental conditions prevailing there and to estimate the total Hg and CH_3Hg^+ levels in the sediments. In addition, bulk samples of surficial bottom sediment, shoreline clay, and sphagnum moss were collected for utilization in laboratory experiments.

Water. Temperature and conductivity profiles and the pH values of surface and bottom water were measured with a temperature-salinity-conductivity meter and pH meter. Water samples were collected for chemical analysis from the surface and within 1 m of the bottom with the aid of a van Dorn bottle. Chemical analyses were performed on both whole and filtered water. Most chemical analyses of water samples were done by the methods of Stainton *et al*. (1977).

Bottom water collected in August, 1983, was analyzed for dissolved oxygen, total and "dissolved" (< 0.45 μm) organic carbon and dissolved iron, and surface water was analyzed for chlorophyll-*a*. Water samples taken in June, 1984, were analyzed for dissolved oxygen, calcium, magnesium, and iron, and for dissolved and suspended organic carbon, nitrogen, and phosphorus, as well as biological oxygen demand (American Public Health Association *et al*., 1985), "soluble reactive phosphorus," and chlorophyll-*a*.

Sediments. Bottom sediments were sampled with an Ekman dredge. The pH and Eh of the sediments were measured with a pH meter (employing calomel and platinum electrodes for Eh readings).

Total Hg analyses were performed by digesting sediment samples with boiling HCl/HNO_3 (*aqua regia*), reducing the dissolved Hg^{2+} to Hg^o with $SnSO_4$, and determining the Hg^o by flameless atomic absorption spectrophotometry (AAS) employing a Perkin-Elmer 403 unit (Mercury Sampling and Analysis Review Committee, 1977). CH_3Hg^+ was extracted from the sediments with $CuSO_4$ and $NaBr/H_2SO_4$ solutions and toluene, and the extracts were purified by $Na_2S_2O_3$ and KI treatments and accompanying polar/nonpolar solvent fractionations, whereupon the CH_3Hg^+ was determined by gas chromatography (GC) using a Tracor Micro-Tek (MT) 220 GC unit with a Varian electron-capture detector (Uthe *et al.*, 1972). The GC analyses were routinely accompanied by confirmation tests employing Ag_2SO_4 to correct for interference (Jensen, 1969). (It should be noted that the only organic (i.e. organometallic) Hg compound other than CH_3Hg^+ which is likely to have occurred in the sediments is $(CH_3)_2Hg$ (dimethyl Hg), which, like CH_3Hg^+, is bio-available and toxic, though it is less efficiently taken up by organisms owing to loss to the atmosphere through volatilization. But even if $(CH_3)_2Hg$ was present in detectable quantities, it was probably transformed spontaneously into CH_3Hg^+ by the acidic reagents used in the extraction process (Decadt, G., Vrije Universiteit Brussel, Brussels, Belgium, personal communication). Thus, CH_3Hg^+ and $(CH_3)_2Hg$ are indistinguishible by the methods employed). Note that in this report "inorganic Hg" is defined as the unmethylated fraction of the Hg (i.e. the difference between the total and methyl Hg concentrations).

The sediments were also analyzed for total iron, manganese, and selenium by AAS techniques following digestion with hot $HNO_3/HClO_4$, and organic carbon and total nitrogen were determined with a Carlo-Erba CHN analyzer. Total nonvolatile sulfide was estimated by digestion with hot 6 N HCl to convert all of the sulfide to H_2S gas, which was then trapped in zinc acetate solution, transformed into a coloured complex by reaction with p-phenylenediamine dihydrochloride and $FeCl_3$, and measured spectrophotometrically (Stainton _et_ _al_., 1977). Hydrated oxides (or "oxyhydroxides") of iron and manganese were extracted sequentially with $NH_2OH{\cdot}HCl/HNO_3$ and citrate/dithionite, in that order, and were estimated as elemental iron and manganese by AAS analysis of the extracts (Holmgren, 1967; Chao, 1972; Gupta and Chen, 1975). Grain-size frequency was determined by wet-sieving to remove sand followed by estimation of the silt and clay fractions with a Sedigraph.

Experimental procedures.

Sediments reserved for utilization as sources of methylating and demethylating microbes in the experiments were stored in plastic bags at 4°C. To the greatest extent possible, air was excuded from the bags. Air slowly diffused through the plastic, causing precipitation of hydrated iron oxide, but only unoxidized material from the centre of each sample was used in the experiments. An attempt was made to exclude air, or at least to slow the rate of oxidation, by storing sample bags under an atmosphere of nitrogen. Sediments collected in August, 1983, were used until June, 1984, when they were replaced by freshly collected sediments; it was not feasible to collect fresh material during the intervening time. Nonetheless, even after months of cold storage, viable microbes capable of methylating or demethylating Hg were invariably present, although the strictly anaerobic methane-producing bacteria became inactive 5 to 6 months after sample collection, and other alterations in the species composition of the microbial communities cannot be ruled out.

Some experiments involved comparative study of the methylating capabilities of sediments from different field sites representing different depositional environments and widely varying degrees of environmental alteration due to flooding. The sediments were incubated under uniform conditions for fixed or varying time spans, and site-related differences in the results were noted. Most experiments, however, were designed to test effects of different physicochemical variables on the methylation or demethylation of Hg by microbes in sets of replicate sediment samples from a single field site. Owing to the limited number of samples that could be handled in any one experiment, it was not feasible, in such experiments, to test sediments from more than one field site; nor was it considered necessary, as the results were deemed to be illustrative of general principles that apply to the field area as a whole. (The validity of this assumption was verified by a comparative study of the effects of nutrient enrichment on Hg methylation in sediments from all 12 of the sampling sites: 10 of the 12 sediment samples showed greatly enhanced rates of methylation on addition of an organic substrate). In most cases, sediment from East Mynarski Lake was used as the source of Hg-transforming microbes; there were two reasons for this choice: (1) the lake is relatively productive and has an ample microflora; and (2) the lake has not been affected by hydroelectric development, so that the experiments performed on samples of its bottom sediments simulated effects of impoundment on a pristine lake. In a few of the experiments, sediment from East Notigi Lake was used instead.

The reason for testing entire communities of microbes in their native sedimentary matrix instead of individual microbial species isolated from the sediments and cultured on synthetic media was that this approach was considered more likely to yield results on which meaningful interpretations and predictions of events occurring in nature could be based. In natural ecosystems, CH_3Hg^+ is synthesized and destroyed by many different types of microbe that differ from one another in their ecological requirements. With changing ambient conditions, ecological succession is to be expected, and a previously dominant species of methylating or demethylating microbe may become inactive while another, previously suppressed species rises to a position of dominance (Jackson, 1984a). The rate of Hg methylation or demethylation in a particular mass of sediment under a given set of environmental conditions is the net result of the activities and interactions of the multitude of different species and strains that make up the microbial community as a whole. In experimental microbial ecology, it makes sense - at least at the outset - to work with entire communities rather than pure cultures of isolated species. The detailed study of single species or strains is the next logical step, but this was not feasible in the present study.

In experiments designed to test effects of selected environmental factors, the factors being tested were varied while all others were kept constant, and comparisons were made between sets of "experimental" and "control" samples. The initial, or baseline, conditions were ascertained by analysis of samples taken at the outset of each incubation cycle. The experimental design conformed to either of two basic patterns: (1) Sets of identical experimental ecosystems, each set differing in one particular respect from the others (for instance, in the composition of its atmosphere), were incubated for varying lengths of time (up to a maximum of

21 days, or, more frequently, 7 or 14 days); or (2) experimental ecosystems representing a range of values for some environmental parameter (for instance, the pH or the quantity of organic substrate added) were all incubated for a single selected length of time (generally 7 to 10 days).

The experimental technique was a modification of the technique developed by Furutani *et al.* (1984). In each experiment, the sediment to be tested was homogenized by stirring with a spatula, and 10 g portions were weighed into 125 mL Erlenmeyer flasks fitted with silicone stoppers, each of which was transfixed by a glass tube with a 3-way stopcock (for sampling head gas) attached to its upper end by a piece of tygon tubing. The sediment in each reaction vessel was mixed with 20 mL of $HgCl_2$ solution (for methylation experiments) or CH_3Hg^+ acetate solution (for demethylation experiments) or, in some cases, plain deionized distilled water, together with whatever additional ingredients were required for the particular experiment, and the composition of the atmosphere in the flask was adjusted as required; the flasks were then incubated for the predetermined period, or periods, of time in a dark chamber at about 25°C. In methylation experiments, the concentration of $HgCl_2$ was 10 μM (except when the concentration of $HgCl_2$ was itself the variable being tested), and in the demethylation experiments the comcentration of CH_3Hg^+ acetate was 100 nM. In most experiments 0.1000 g (or, in a few experiments, 0.5000 g) of air-dried sphagnum moss pulverized with a Cyclone sample mill (UD Corp.) was added to each flask as a nutrient substrate (sphagnum moss being a widespread and abundant component of the terrestrial vegetation in northern Manitoba) (cf. Hecky *et al.*, 1986). In some experiments, varying amounts of moss were added, or moss was added to some flasks but not others.* Single-celled green and blue-green algae (Scenedesmus quadricauda and Anabaena variabilis, respectively) which had been grown in batch cultures and then freeze-dried were used as nutrient substrates in two of the experiments, 0.02 g of algal cells being added to each reaction flask.

Initially the experiments were performed without the addition of a pH buffer so that the environmental conditions would be as close as possible to those existing in nature, but appreciable variations in pH were commonly observed by the end of the incubation period; therefore, many of these experiments were repeated with the addition of 1 g of reagent-grade calcium carbonate ($CaCO_3$) to each flask at the outset in order to maintain uniform, approximately neutral pH conditions and rule out the possibility that previously observed experimental results were due to pH effects. Although pH was indeed found to influence the results, essentially the same conclusions were reached regardless of whether the slurries were buffered with $CaCO_3$. In one experiment designed to test the influence of pH on methylation and demethylation, the initial pH values were adjusted to the range 4-10 by means of phthalate, phosphate, tris(hydroxymethyl)-aminomethane, and borax buffers.

The atmosphere of each reaction vessel was nitrogen (N_2), air, or hydrogen sulfide (H_2S). In all experiments other than those expressly designed to test effects of varying oxidation-reduction conditions, each sediment-water slurry was incubated under an atmosphere of commercial grade

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*Chemical analysis of the moss revealed that its mean Hg content was only 0.175 μg/g. Therefore, the amount of Hg unintentionally added with the moss was generally three orders of magnitude less than the amount added in the form of $HgCl_2$.

(99.9% pure) N_2 (often with an oxygen-removing device on the gas line), as exclusion of air was found to foster methylation and demethylation of Hg.

In a number of experiments the sedimentary environment was varied by addition of certain natural and synthetic inorganic solids, including shoreline clay (Pleistocene glacio-lacustrine material from the shore of Southern Indian Lake) (cf. Hecky _et al_., 1986) as well as the pure clay minerals kaolinite ("colloidal kaolin," K6, Fisher Scientific Co.) and montmorillonite ("bentonite," B-235, Fisher Scientific Co.), synthetic ferric oxyhydroxide (FeOOH) and manganese dioxide (MnO_2) prepared by the procedures of Kinniburgh _et al._ (1975) and Lockwood and Chen (1973), and reagent-grade ferrous sulfide (FeS). Grain-size analysis performed by wet-sieving and centrifugation techniques showed that the shoreline clay was "silty clay" finer than 4 phi; X-ray diffractometry revealed that it is a mixture of clay minerals, quartz, feldspars, and carbonate minerals (calcite and dolomite), and that its clay fraction is composed mainly of illite, with smaller amounts of kaolinite, chlorite, and expanding-lattice clay. The shoreline clay was either used in its native state or subjected to different treatments before being used for experimental purposes. The pre-treatments comprised (1) dialysis (for removal of adsorbed substances), (2) suspension in 1 N $CaCl_2$ solution (for removal of adsorbed substances and creation of a uniform ionic environment at the exchange sites), followed by dialysis (for removal of excess $CaCl_2$ and the desorbed substances) (3) exposure to 1 N HCl followed by dialysis (to remove carbonates together with labile oxide coatings and other adsorbed substances), (4) digestion in boiling 30% H_2O_2 (to destroy organic matter), followed by $CaCl_2$ treatment and dialysis, and (5) citrate/dithionite treatment of H_2O_2-treated clay (to remove iron/manganese oxide coatings)

(Holmgren, 1967) followed by $CaCl_2$ treatment and dialysis. The kaolinite and montmorillonite were treated with 1 N $CaCl_2$ solution and then dialyzed, and the synthetic oxides were dialyzed before being dried. All dialysis operations were performed with a Pope multiple dialyzer, and the progress of dialysis was monitored with a conductivity meter or pH meter.

All experiments were performed under sterile conditions, although it was not feasible to collect the sediments aseptically in the field. Glassware, stoppers, and other equipment, together with all solid ingredients of the sediment-water slurrries (except the sediment itself) were autoclaved, whereas the $HgCl_2$ and CH_3Hg^+ solutions were sterilized by filtration with autoclaved 0.45 μm (or, in a few cases, 0.7 μm) Millipore filters. A few experiments employing autoclaved sediment were done in order to check for abiotic production or breakdown of CH_3Hg^+.

After the pre-determined period of incubation, the experimental data were obtained by the following techniques: (1) Determination of the concentrations of carbon dioxide (CO_2) and methane (CH_4) in the head gas, which was sampled with a syringe and analyzed with a "Basic" GC unit (CA Instruments, model 8000), (2) measurement of the pH and Eh of the slurry with a Corning 12 pH meter, using calomel and platinum electrodes for the Eh measurements, (3) determination of the CH_3Hg^+ content of the slurry using the methods of solvent extraction, GC analysis, and confirmation with Ag_2SO_4 described above, and (4) determination of the total Hg content of the slurry (as required), using the methods of wet-ashing and flameless AAS referred to previously. The CH_3Hg^+ and total Hg concentrations were recalculated as the absolute quantities of CH_3Hg^+ and total Hg in each slurry so that meaningful comparisons could be made between data for slurries containing different amounts of solid ingredients.

In addition to the methylation and demethylation experiments, some experiments on the adsorption of inorganic bivalent Hg by shoreline clay, pure clay minerals, and oxides were carried out. Varying quantities of these materials were suspended in 20 mL of 10 µm $HgCl_2$ for about 24 hours. the slurries were then centrifuged, and the solution phase of each sample, together with a control solution of $HgCl_2$ containing no suspended clay or oxide, was digested with hot HNO_3/H_2SO_4 and $KMnO_4 + H_2O_2$ (Mercury Sampling and Analysis Review Committee, 1977) and then analyzed for total Hg by flameless AAS.

Finally, the biogeochemical processes regulating the accumulation of Hg by fish in different lakes of the field area were investigated by exploring the relationships between the following sets of data: (1) Hg concentrations in muscle tissue of fish caught in 1980 and 1981 (Bodaly _et al._, 1984a; McGregor, G.W.G., unpublished data, personal communication); (2) the Hg methylating capability, or potential methylation rate, of bottom sediment as determined by a laboratory bioassay in which added inorganic Hg(II) was transformed to CH_3Hg^+ by the sediment's microflora under standardized conditions; (3) concentrations of CH_3Hg^+ and total Hg in the sediments under natural conditions, as determined by analysis of freshly collected field samples (Jackson, Ms. in preparation); and (4) values for three different parameters which, in the writer's opinion, are indexes of the abundance of available inorganic Hg species in the sediments - "available inorganic Hg" being defined as unmethylated Hg species that are susceptible to methylation and bio-accumulation, presumably because they have not been immobilized by Hg-binding agents such as sulfides and selenium species. The "methylating capability" of the sediment was estimated as the quantity of CH_3Hg^+ generated per unit weight of sediment upon incubation of sediment samples under N_2 at 25^oC. for 10 days in the presence of added $HgCl_2$ solution without nutrient enrichment (see above). The "available inorganic Hg" fraction of the sediment was represented by the concentration of weakly bound, exchangeable Hg species, defined as all forms of Hg that were extractable with a 1 N $CaCl_2$ solution, and by the molar ratios total Hg/sulfide and total Hg/selenium (symbolized as Hg/S^{2-} and Hg/Se, respectively). Because organic and inorganic sulfides and selenium species were considered the sediment constituents most likely to interfere with the methylation and biological uptake of Hg owing to their exceptionally strong tendency to form stable covalent bonds with Hg, the writer conjectured that the availability of Hg would tend to correlate with the Hg/S^{2-} or Hg/Se ratio. The $CaCl_2$-extractable Hg concentrations and Hg/S^{2-} ratios represent samples collected in August, 1981; these data were taken from earlier work by Jackson (Ms. in preparation).

The exact forms of the Hg in the "available inorganic Hg" fraction are, of course, unknown, but the tentative assumption that they are predominantly inorganic (unmethylated) is consistent with the following observations: (1) paired sets of CH_3Hg^+ and $CaCl_2$-extractable Hg data obtained for sediment samples from different lakes did not correlate significantly with each other, indicating that they are separate and distinct Hg fractions; (2) comparison of CH_3Hg^+ and total Hg levels in sediments revealed that the total Hg consists mainly of unmethylated Hg (as would be expected), signifying that the forms of Hg represented by the Hg/S^{2-} and Hg/Se ratios are predominantly inorganic; and (3) $CaCl_2$-extractable Hg and the Hg/S^{2-} and Hg/Se ratios all gave essentially the same results and thus were found to be virtually interchangeable as indexes of availability (see below), suggesting that all three parameters represent the same pool of Hg species; consequently, the fact that the Hg/S^{2-} and Hg/Se ratios represent inorganic forms of Hg implies that $CaCl_2$-extractable Hg is mostly inorganic as well.

RESULTS AND DISCUSSION

Comparison of research data for different field sites.

Analytical data for field samples from Granville Lake and Methyl Bay (Southern Indian Lake). The analytical data demonstrate strikingly that the flooding of densely forested land bordering the riverine lakes has resulted in abnormally high concentrations of CH_3Hg^+ and total Hg in bottom sediments, as well as greatly elevated CH_3Hg^+/total Hg ratios (suggesting elevated rates of CH_3Hg^+ production from inorganic Hg), at the reservoir sites relative to the control site, the highest values for these parameters occurring directly on top of flooded land (Fig. 2). Thus, the CH_3Hg^+ and total Hg concentrations and CH_3Hg^+/total Hg ratio increase in the order Granville Lake < Methyl Bay, offshore < Methyl Bay, flooded forest. This is also true of the inorganic Hg concentration, which closely approximates the total Hg concentration (inorganic Hg being 1 to 2 orders of magnitude more abundant than CH_3Hg^+). Similarly, the organic carbon content of bottom sediment increases, while the dissolved oxygen content of bottom water and the Eh of bottom sediment decrease, in the same order (Fig. 3).

These observations support following hypotheses: (1) Flooding has caused a marked increase in rates of Hg methylation in the bottom sediments, and the rates tend to correlate with the magnitude of the environmental alteration due to flooding. (2) The elevated rates of methylation have resulted primarily from enhancement of microbial activity owing to the environmental alterations; the evident increase in the supply of unmethylated, or "inorganic," Hg, which is probably derived from scattered sources in submerged terrestrial organic materials, is a contributing factor but is of secondary importance. (3) The elevated rates of CH_3Hg^+ production have resulted from the stimulation of anaerobic heterotrophic bacteria by the introduction of labile organic matter from flooded land. (4) Promotion of the activities of methylating microbes by this organic matter is due not only to nutrient utilization by these microbes but also to oxygen depletion caused by the activities of aerobic heterotrophic microbes. The results of subsequent experimental work strongly support these inferences (see below).

Experimental data: Results of methylation assays of sediments from Granville Lake and Methyl Bay (Southern Indian Lake). The analytical results for the sediment samples were confirmed by methylation assays performed under laboratory conditions.

In assays of sediment samples collected in both 1983 and 1984, the quantities of CH_3Hg^+ produced from $HgCl_2$ upon incubation for 7 to 14 days under N_2 without added moss were consistently 2 or 3 orders of magnitude higher for the flooded-forest zone of Methyl Bay than for the offshore zone of Methyl Bay and the Granville Lake site (both of which sites gave nearly the same results) (Figures 4 and 5). Similarly, sediments from the flooded-forest site generated greater quantities of CO_2 (and CH_4 in the case of freshly collected 1983 sediment), indicating higher levels of microbial activity (Figures 4 and 5). Evidently the flooded-forest environment fosters the growth of CH_4-producing bacteria (Fig. 4), presumably owing to the abundance of decomposing plant remains in that region, and, under anoxic conditions, these bacteria may well contribute to the methylation of Hg by means of a biochemical pathway linked to the pathway for CH_4 production (Wood et al., 1968; Jackson, 1984a). Nevertheless, CH_4-producing bacteria are not essential to the methylation process. Vigorous Hg methylation also occurred in the absence of

CH_4-producing activity, though in the presence of CO_2-producing activity, demonstrating that CH_3Hg^+ is synthesized by microbial species other than CH_4-producing bacteria. After 6 months' storage at 4°C., Methyl Bay sediment from the flooded forest zone had lost its ability to generate CH_4 but retained its exceptionally strong Hg methylating capability (Fig. 5A). However, in the total absence of viable microbes (when the sediments were autoclaved prior to incubation), CH_3Hg^+ production was completely suppressed, verifying the assumption that microbial activity is indispensible to the methylation process.

Experimental data: Results of methylation assays of sediments from all field sites. A survey of the methylating capabilities of sediments from all 12 of the field sites sampled in 1983 (incubated for 10 days under N_2 without added moss) revealed an extremely significant positive exponential correlation between the quantities of CH_3Hg^+ and CO_2 generated per unit weight of sediment (Fig. 6A). The CO_2, in turn, gave a strong positive linear correlation with the organic carbon content of the sediment (Fig. 6B). The flooded forest site of Methyl Bay showed the highest CH_3Hg^+, CO_2, and organic carbon levels. Among the offshore sites, the rate of CH_3Hg^+ production increased in the order Granville Lake $<$ Southern Indian Lake $<$ Notigi Reservoir - that is, in the order of increasing rise in lake level due to the Churchill-Nelson diversion (Fig. 7).

The relationships illustrated in Figures 6 and 7 strongly support the conclusion that flooding caused by the diversion has resulted in a very pronounced increase in rates of Hg methylation, and that this increase is due to stimulation of heterotrophic microbial activity by labile organic matter (mostly decomposing remains of plants) in the submerged land areas. Effects of different environmental factors on microbial methylation and demethylation of Hg: Experimental results.

Organic nutrients and oxidation-reduction effects. Since trophic conditions and oxidation-reduction regime are interrelated, they are treated together in this section.

Addition of pulverized moss to the sediment-water slurries promoted the methylation of inorganic bivalent Hg under both aerobic and anaerobic conditions (Figures 8 and 9). Although methylation occurred in both the presence and the absence of oxygen, the rate of CH_3Hg^+ production was much higher when oxygen was excluded (Figures 8 and 9), provided the sulfide concentration was low. FeS tended to inhibit CH_3Hg^+ production, and abundant H_2S suppressed it altogether (Fig. 10), probably (at least in part) by precipitating the Hg^{2+} as HgS, thereby rendering it unavailable for methylation (Jernelöv, 1972). Note that H_2S did not suppress CO_2 production, apparently indicating that the inhibitory effect of H_2S on methylation cannot be explained by inhibition of heterotrophic microbial activity as a whole.

Interestingly, in the experiment represented by Figure 10 the sediments incubated under N_2 without added FeS were the only ones that generated CH_4, besides generating the largest quantities of CH_3Hg^+. Evidently the methanogenic bacteria were sensitive to sulfides as well as to oxygen. The data are consistent with the reasonable assumption that the production of CH_3Hg^+ is accomplished by different microbial species under different environmental conditions - for instance, aerobic or microaerophilic methylators in the presence of free oxygen, and anaerobic or facultatively anaerobic methylators in anoxic (but not extremely reducing) environments.

Utilization of killed cells of different kinds of freshwater algae as nutrient substrates gave results comparable to those obtained with moss. Thus, algal organic matter promoted the methylation of Hg (Fig. 11), and from the standpoint of quantitative yield it made practically no difference whether Anabaena or Scenedesmus was used as the substrate, even though there is a profound phylogenetic difference between these two types of alga.

To summarize briefly, various organic substrates such as occur naturally in the flooded forest land and lake water strongly promote CH_3Hg^+ production by different kinds of heterotrophic micro-organism under both aerobic and anaerobic conditions. Anaerobic environments are more favourable than aerobic environments, provided the sulfide levels are low, as is generally the case in the lakes and reservoirs studied (the concentrations of sulfide in sediments ranging from undetectable to 0.64 µg/g in all sampling regions except East Mynarski Lake, whose sediments gave a concentration of 48.5 µg/g); abundant sulfide, such as occurs under highly reducing conditions, tends to inhibit methylation. In general, labile organic matter has the two-fold effect of furnishing nutrient substrates for microbes and creating anaerobic or oxygen-poor conditions. Thus, terrestrial organic matter - especially remains of plants - introduced into aquatic environments of northern Manitoba as a result of impoundment and flooding has greatly enhanced CH_3Hg^+ production by (1) furnishing nutrients for methylating microbes and (2) depleting oxygen levels without generating enough sulfide to cause net inhibition of Hg methylation.

Now let us consider the possible role of demethylation processes. The CH_3Hg^+ content of replicate $HgCl_2$-spiked slurries incubated for different periods of time typically showed an initial upsurge, then peaked and declined steadily with further incubation, or levelled off, while microbial activity (as indicated by the CO_2 and CH_4 data) continued unabated (Figures 9, 10, and 12). This constitutes evidence that methylation is normally counterbalanced to a greater or lesser degree by demethylation. A particularly striking and instructive example involving comparison of results obtained with and without enrichment of the slurry with moss is shown in Figure 12. In the presence of moss, CH_3Hg^+ and CO_2 both increased progressively for the first 10 days of incubation; after the 10th day, CH_3Hg^+ levels declined steadily, while CO_2 levels started to fall but then rose again after the 14th day. The drop in CH_3Hg^+ levels following the 10th day coincided with a sharp, continuous rise in CH_4 production. These data appear to indicate that methylating bacteria predominated for the first 10 days of incubation and that they were subsequently superceded by demethylating bacteria. The variations in CO_2 and CH_4 levels with time support the conclusion that the methylation and demethylation of Hg were catalyzed by two distinctly different species or communities of microbe. The data suggest that the demethylators were methanogenic bacteria. Recalling that in some experiments CH_4 production appeared to be linked to methylation, we may reasonably postulate that methanogenic bacteria are capable of mediating both the synthesis and the breakdown of CH_3Hg^+. Perhaps both reactions are side-effects of the biochemical pathway for CH_4 production. Possibly methyl groups can be generated at the expense of CH_4, or CH_4 at the expense of methyl groups, depending on the environmental conditions or the particular strain of methanogenic bacteria that is active at the time. More work is needed to ascertain the role, if any, of methanogenic bacteria.

In the absence of moss, CH_3Hg^+ concentration increased initially but quickly levelled off and remained virtually constant for the remainder of the incubation period. Either methylation ceased soon after it began, and the CH_3Hg^+ produced was not broken down by demethylators, or a state of dynamic equilibrium between methylation and demethylation was rapidly established. The second possibility seems more plausible, as the CO_2 concentration increased progressively with time throughout the experiment, indicating continuous heterotrophic microbial activity; under such circumstances, total cessation of both methylation and demethylation would be unlikely.

Figures 9, 10, and 12 show the results obtained when sediment mixed with a fixed amount of moss was incubated for different lengths of time. But what happened when the quantity of moss was varied and the incubation time was kept constant? We have seen that quantities of moss on the order of 0.005-0.05 g stimulated methylation (Fig. 8), but an unlimited increase in the amount of moss did not sustain an unlimited increase in CH_3Hg^+ level. In one experiment, in which sediment was incubated under N_2 in the presence of 0.005-0.5 g of moss, CH_3Hg^+ production increased as the quantity of moss increased up to 0.2 g of moss per flask, but decreased with further increases in the quantity of moss, while CO_2 increased steadily and the Eh declined steadily with increasing moss over the entire range of substrate levels (Fig. 13). These observations suggest that methylation was superceded by demethylation at the highest concentration of moss. Methylation assays of sediments from different field sites in the presence and absence of added moss led to the same conclusions: Addition of moss actually depressed CH_3Hg^+ production by sediments from the flooded forest zone of Methyl Bay and from East Mynarski Lake, the two field sites where the sediments had the highest organic carbon concentrations and the highest potential for CH_3Hg^+ and CO_2 production (without artificial fertilization); but moss caused large increases in the CH_3Hg^+ yield of sediments from the other sites (including Granville Lake and the offshore zone of Methyl Bay), whose sediments had lower concentrations of organic carbon and lower CH_3Hg^+ and CO_2 producing capabilities to begin with (Fig. 14, A and B).

Demethylation assays of sediment in the presence and absence of moss and under different oxidation-reduction regimes yielded the data displayed in Figure 15, in which the CH_3Hg^+ content of the slurry is plotted against time. Comparison of these results with the results of methylation assays performed under the same conditions (Fig. 9) revealed both similarities and differences in the effects of nutrients and oxidation-reduction regime on the activities of the microbes. Under an atmosphere of air, fertilization of the sediment with moss induced a substantial increase in the rate of demethylation; but under an atmosphere of N_2, moss depressed the demethylation rate somewhat. When moss was added, the rate of demethylation was nearly the same regardless of whether the atmosphere was air or N_2 (perhaps because the moss created anaerobic micro-environments even in the presence of free oxygen), although air had a slightly more favourable effect than N_2. Thus, N_2 with no moss furnished the most favourable environment for demethylation, while air with no moss furnished the least favourable environment, and air or N_2 with added moss was intermediate in its effects. Quantitatively, the greatest difference was between the moss-free aerobic environment and all of the others together. Incidentally, it should be borne in mind that the curves plotted in Figure 15 may represent the net effect of competing demethylation and remethylation processes, as some of the inorganic Hg generated by demethylating microbes may well have been converted back to CH_3Hg^+ by methylating microbes.

To summarize, the experimental results displayed in Figures 9 and 15 showed that the N_2 + moss environment was ideal for methylation but was also relatively favourable (though not optimal) for demethylation, and air without moss was unfavourable for both methylation and demethylation, while air + moss and N_2 without moss - in particular the latter - were more or less unfavourable for methylation but favourable for demethylation. Whether environmental conditions in reservoirs could, or should, be manipulated deliberately so as to promote demethylation at the expense of methylation is highly questionable, and more research would have to be done - preferably under field conditions - before any decisions were reached; but the experimental results suggest that this approach to the problem of amelioration merits further study. Perhaps a more satisfactory approach, however, would be to prevent future environmental damage in what remains of Canada's unspoiled land and inland waters by profiting from the expensive mistakes made in northern Manitoba and elsewhere. To say the least, the introduction of large masses of labile organic matter into any body of water where edible fish are of major concern should be avoided.

Inorganic bivalent Hg concentration. Incubation of sediment under N_2 with and without added moss for a fixed period of time in the presence of different concentrations of dissolved $HgCl_2$ ranging from zero to a maximum of 1,000 µM (which gave a concentration in the slurry 5 orders of magnitude greater than that found in any of the sediment samples) yielded the results illustrated in Figure 16 (A and B). The CH_3Hg^+ yield increased with increasing $HgCl_2$ concentration, but in slurries fertilized with moss CH_3Hg^+ production peaked at 100 µM $HgCl_2$ and then declined, presumably because of toxic effects of the highest $HgCl_2$ concentration on the methylating microbes (cf. Jackson, 1984a). In slurries not fertilized with moss the CH_3Hg^+ yield, though generally smaller, increased progressively over the entire range of $HgCl_2$ levels. In the presence of moss, CH_4 was generated, and the CH_4 levels correlated quite closely with those of CH_3Hg^+, suggesting (though not proving) that the CH_3Hg^+ was synthesized mainly by methanogenic bacteria; in the absence of moss, however, no CH_4 was detected, and so CH_3Hg^+ synthesis was evidently accomplished by other types of microbes. Apparently the dominant Hg methylating microbes in the nutrient-enriched slurries were more sensitive to toxic effects of inorganic bivalent Hg than were the methylators in the slurries which had not been enriched.

According to these findings, any input of inorganic bivalent Hg from natural accumulations in flooded land areas can be expected to foster an increase in CH_3Hg^+ production, the rate of production varying widely, however, with trophic conditions.

Ambient pH. Assays conducted in the pH range 4-10 under atmospheres of N_2 and air in the presence of moss demonstrated that pH values of approximately 6.0-7.5 were optimal for both methylation and demethylation (except for aerobic methylation, whose optimal pH was close to 8 (Fig. 17). At $pH < 6$, methylation and demethylation occurred at the same rates in air as in N_2; but at $pH > 6$ there were large differences between the results obtained under air and N_2: In the pH range 6-8 methylation rates were higher in N_2 than in air, whereas in the pH range 6-9 demethylation rates were higher in air than in N_2 (Fig. 17). Note that in both the methylation and the demethylation experiments performed under anaerobic conditions, CH_4 was generated only where CH_3Hg^+ production and decomposition, respectively, were greatest (within a pH range of about 6.6-8.1), suggesting, again, that methanogenic bacteria play a significant role in both methylation and demethylation. In both experiments, maximum CO_2 production occurred at pH 6.1-6.2, and thus did not coincide with CH_4 production.

According to these results, the pH conditions prevailing in the field area are ideally suited to both methylation and demethylation, the mean pH values and pH ranges recorded for sediment and water samples collected from the 12 field sites in 1983 being as follows: Sediments, pH 6.93 (6.60-7.25); bottom water, pH 6.99 (6.50-7.30); surface water, pH 7.36 (6.70-8.50). The applicability of the experimental laboratory data to field conditions is uncertain, but the data suggest that an abundance of dissolved oxygen and an ambient pH in the range 6-8 are desirable from the standpoint of promoting demethylation at the expense of methylation. On the other hand, moderately acidic pH levels (say, in the range 5-6) and oxidation-reduction conditions ranging from aerobic to anaerobic might be equally desirable, as suggested by Figure 18, which compares rates of CH_3Hg^+ production by selected sediment samples in the presence and absence of the buffering agent $CaCO_3$. The $CaCO_3$ raised the pH by approximately one pH unit (from a range of 5.81-6.75 to a range of 6.87-7.23), which was enough to cause a considerable rise in the rate of methylation. The effect of buffering on the CH_3Hg^+ yield was especially striking in the case of sediment from East Mynarski Lake.

The role of shoreline clay in pH buffering is treated in the next section.

Clay minerals and oxides. The environmental effects of Pleistocene glaciolacustrine clay eroded into Southern Indian Lake include pH buffering by carbonate minerals (calcite and dolomite) and other effects that are apparently unrelated to the neutralization of acids. Possibly the diversity and complexity of the clay's ecological functions explains the seemingly inconsistent results obtained by other workers in limnocorral experiments dealing with the role of clay in the methylation and bioaccumulation of Hg (Hecky *et al.*, 1986).

Methylation assays showed that shoreline clay in its native state, or merely dialyzed without removal of carbonate minerals, caused increased production of CH_3Hg^+ by sedimentary microbes (Figures 19 and 20). In the absence of any artificial buffer, the clay raised the pH of the slurry from about 6 to as much as 7.43 (depending on the quantity of clay added) (Figures 19 and 20), and, in view of the pH effects discussed in the previous section, it is reasonable to conclude that the stimulation of Hg methylation by the clay resulted from pH buffering by carbonate minerals in the clay rather than effects of the clay minerals and silt particles. This interpretation is supported by the results of a subsequent experiment patterned after the one represented by Figure 20 except that all experimental and control slurries were buffered with $CaCO_3$, thus preventing the pH variations observed in the previous experiments. The outcome (Fig. 21) indicated that when the pH effect was thereby eliminated the clay *inhibited* CH_3Hg^+ production slightly (although the effect was so small that its significance is open to question).

Figures 20 and 21 are worth closer inspection, as they have several interesting though puzzling features meriting further research. In both the buffered and the unbuffered slurries, and in the presence and absence of dialyzed but otherwise untreated clay, the CH_3Hg^+ level increased with time for the first 7 days of incubation. For the next 7 days, the CH_3Hg^+

content of the controls (the slurries without added shoreline clay) levelled off (Fig. 20) or declined gradually with time (Fig. 21); in contrast, the CH_3Hg^+ concentrations in both buffered and unbuffered slurries containing clay declined sharply and dramatically with time (the phenomenon being especially striking in the buffered slurries). In both buffered and unbuffered slurries, clay pretreated with acid strongly depressed the production of CH_3Hg^+, but the CH_3Hg^+ concentrations, having reached their maximum, declined only slightly thereafter, as in the control slurries (Figures 20 and 21). In both experiments the CO_2 levels increased initially and then declined with further incubation (or declined and then came back up) (Figures 20 and 21). Production of CO_2 was consistently greatest in the presence of acid-treated clay, apparently ruling out the possibility that inhibition of CH_3Hg^+ production by acid-treated clay was due to inhibition of heterotrophic microbial activity (release of CO_2 from the $CaCO_3$ buffer could have been a contributing factor in the case of the results shown in Figure 21, but this would not explain the eventual decline in CO_2 levels or the close parallels between Figures 20 and 21). Indeed, the acid-treated clay seems to have stimulated CO_2-generated microbial activities even while depressing the net production of CH_3Hg^+. The drop in CO_2 levels following the initial increase probably resulted from heterotrophic CO_2-fixation reactions (such as biosynthesis of malic acid from pyruvic acid, for example), and suggests the rise of a previously inactive or less active type of microbe to a position of dominance. In the slurries containing unacidified clay (and to a lesser extent in the other slurries), the variations in CO_2 concentration tended to correspond to the variations in CH_3Hg^+ level, suggesting that the sharp drop in CH_3Hg^+ level after the 7th day was due to an increase in the growth of demethylating microbes which may have possessed one or more of the biochemical pathways for heterotrophic reduction of CO_2. On the other hand, acid-treated clay suppressed demethylation but promoted the growth of CO_2-fixing microbes.

The reason for the radically different results produced by acid-treated and unacidified clay can only be guessed without additional information. Alteration of the surface properties of the clay owing to solubilization of oxide coatings and the mobilization of aluminum in the octahedral layer is a likely effect of acid treatment, and this could account for the contrasting effects of acid-treated and unacidified clay on microbial Hg transformations. Measurement of adsorption capabilities revealed that the acid-treated clay adsorbed more $HgCl_2$ per unit weight of clay than did the unacidified clay; but the relevance of this observation is doubtful, because the adsorption test indicated that the concentrations of clay in the experimental slurries were so high that 100% of Hg^{2+} could well have been adsorbed by both the acid-treated and the unacidified clay. In theory, however, a difference in the strength of adsorption could have played a key role in the subsequent biogeochemical transformations of the Hg^{2+}. It is also conceivable that alteration of the surface chemistry of the clay by acid treatment (for instance, saturation of the exchange sites with Al^{3+}, $Al(OH)^{2+}$, $Al(OH)_2^+$, and H^+ ions) had a toxic effect on the methylating microbes or resulted in the poisoning of the biochemical apparatus for methylation of Hg.

To summarize, the observed effects of shoreline clay - other than the buffering action of its accompanying carbonate minerals - are consistent with the following conclusions: (1) Clay in its native state (dialyzed but otherwise untreated) caused very slight inhibition of Hg methylation, but it greatly promoted subsequent demethylation. (2) Acid-treated clay strongly inhibited Hg methylation and suppressed, or failed to promote, subsequent demethylation of what little CH_3Hg^+ had been synthesized. (3) These effects of untreated and acid-treated clay cannot be ascribed to changes in the bulk pH of the slurry, as they occurred in the presence of added $CaCO_3$, which maintained uniform pH conditions close to neutrality (although pH effects in micro-environments surrounding the clay particles cannot be ruled out). (4) Accompanying CO_2 variations indicate microbial action rather than abiotic chemical reactions catalyzed by the clay minerals.

Further research is required to explain these findings, but the data obtained thus far seem to demonstrate that clay minerals affect the biogeochemical transformations of Hg in a number of different ways and can preferentially promote or inhibit certain microbial species or processes.

In Figures 22 and 23 are given results of experiments designed to determine effects of the organic constituents and oxide coatings of shoreline clay on the methylation and demethylation of Hg. The tests were performed on clay in three different states: (1) "untreated," (2) "H_2O_2-digested," and (3) "citrate/dithionite-extracted" (i.e. H_2O_2-digested and then citrate-dithionite-extracted). All three subgroups of clay had been flooded with $CaCl_2$ solution, dialyzed, and gently air-dried before being used in the methylation and demethylation assays. The experiments were performed under N_2 in the presence of moss and $CaCO_3$. "Control" slurries without added clay were included.

Untreated and H_2O_2-digested clay had little or no effect on methylation, but citrate/dithionite-extracted clay inhibited methylation, causing a 58% drop in CH_3Hg^+ production with respect to CH_3Hg^+ production in the control slurries by the 7th day (Fig. 22). This appears to indicate that the oxide coatings removed by the citrate/dithionite treatment tend to foster the production of CH_3Hg^+, either by playing a direct and positive role in the process or by masking certain sites on the clay surfaces which would otherwise interfere. CO_2 production was highest, and the Eh lowest, in the presence of citrate/dithionite-extracted clay (Fig. 22), apparently signifying that the treated clay, far from suppressing microbial growth as a whole, actually enhanced it. Citrate/dithionite-treated clay depressed demethylation (Fig. 23) as well as methylation, and to nearly the same degree (53% by the 7th day), whereas untreated and H_2O_2-digested clay had little influence on demethylation. Again, CO_2 production was greatest in the presence of citrate/dithionite-extracted clay, while the Eh values were within the range of values given by the other experimental systems (Fig. 23).

The effects of citrate/dithionite-treated clay were comparable to those of acid-treated clay (Figures 20 and 21). Therefore, the results are consistent with the hypothesis that the negative effects of acid-leached clay on methylation and demethylation are attributable to removal of oxide coatings from the clay. On the other hand, the failure of untreated clay to influence demethylation of Hg (Fig. 23) is ostensibly inconsistent with the data displayed in Figures 20 and 21, which apparently show that untreated clay had a large positive effect on demethylation. One possible reason for the apparent discrepancy is that the microbial species responsible for demethylation were not the same in the experiments represented by Figures 20 and 21 as in the experiment represented by Figure 23, and consequently were not affected in the same way by the clay. Clearly further research is needed to clarify the effects of the clay.

Whatever the true explanation for the observed effect of citrate/dithionite-treated clay on methylation and demethylation, it evidently has nothing to do with adsorption of Hg^{2+} by the clay. Experimental study of the adsorption of $HgCl_2$ by shoreline clay demonstrated virtually no difference between the results for untreated, H_2O_2-treated, and citrate/dithionite-treated clay.

The clay minerals kaolinite and montmorillonite, and the oxides FeOOH and MnO_2, had, qualitatively, the same effect as Pleistocene shoreline clay on the kinetics of CH_3Hg^+ production and subsequent destruction, as demonstrated by a methylation assay conducted over a 2-week period under N_2 in the presence of moss and $CaCO_3$ (Fig. 24; cf. Fig. 21). (Note: Data for slurries containing MnO_2 were not included in Figure 24, because these slurries had anomalously high pH values (between 8 and 9)). The following patterns of variation were observed: (1) In both the control slurries and slurries mixed with mineral additives, an initial methylation phase was followed by a demethylation phase, the CH_3Hg^+ reaching its highest concentration before the 7th day and then rapidly declining to levels below the limit of detection. By the 10th day, there was no detectable CH_3Hg^+ remaining in any of the slurries (although a subsequent revival of methylating activity was observed on the 14th day in the control slurry and kaolinite slurry, suggesting growth of a previously dormant or suppressed species of methylating microbe). (2) The clay minerals had no effect on CH_3Hg^+ production during the initial methylation phase (although FeOOH seems to have depressed methylation slightly), but the clay minerals and oxides strongly promoted subsequent demethylation (after the first day of incubation), with the result that demethylation went to completion sooner in the slurries containing added minerals than in the control slurries.

The time required for demethylation to go to completion decreased in the following order: control > kaolinite > montmorillonite = FeOOH > MnO_2. In other words, the tendency of the minerals to accelerate demethylation decreased in the order MnO_2 > FeOOH = montmorillonite > kaolinite. These differences do not seem to be closely related to the ability of the minerals to adsorb Hg^{2+} or to the effects of the minerals on heterotrophic microbial activity. In all experimental environments, CO_2 levels generally increased with time, and the quantity of $HgCl_2$ adsorbed per unit weight of mineral increased in the order montmorillonite < kaolinite < FeOOH < MnO_2*.

The experiments discussed in this section demonstrated effects of clay and associated oxides and carbonates on the formation and decomposition of CH_3Hg^+ under controlled laboratory conditions; but what is the overall effect of clay and silt introduced into lacustrine environments by shoreline erosion and fluvial transport under natural conditions? Perhaps this question can be answered by comparing, once again, the results of methylation assays and analyses performed on sediment samples from different field sites. In assays involving incubation for 10 days under N_2 without nutrient enrichment, South Bay sediment yielded less CH_3Hg^+ than any of the other reservoir samples, and among the Notigi Reservoir samples Rat Lake yielded the least amount of CH_3Hg^+ even though its post-impoundment rise in water level was second only to that of Notigi Lake

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*The observation that adsorption of $HgCl_2$ per unit mass of clay was greater for kaolinite than for montmorillonite was surprising. Normally the opposite would have been expected because of montmorillonite's generally finer grain size, greater specific surface, and higher exchange capacity. To explain this anomaly, it would be necessary to do further work involving comparison of the grain-size frequencies, exchange capacities, and surface coatings of the clay mineral specimens employed in the experiments.

(Table 1). This result can be tentatively ascribed to the anomalously great abundance of eroded shoreline clay and silt in South Bay and the fact that Rat Lake is the receiving basin for clay and silt-bearing water draining out of South Bay (Fig. 1). The geographical distribution of the data suggests that the net effect of detrital clay and silt is to depress the rate of CH_3Hg^+ formation. This can be explained by dilution and rapid burial of organic substrates and other nutrients as well as bacteria by the mineral detritus, as confirmed by the fact that South Bay and Rat Lake sediments were relatively low in organic carbon and nitrogen (Table 1), and by the previously mentioned dependence of methyl mercury production on the sediment's general level of heterotrophic (CO_2 generating) activity, which is related to the concentration of organic substrates in the sediment. Stimulation of Hg demethylation by the clay may be a contributing factor, as suggested by the experimental findings. On the other hand, the buffering action of carbonates in the glacio-lacustrine silty clay evidently has no relevance to the geographical variations in rates of Hg methylation: No pH anomaly was detected in South Bay despite the anomalously great abundance of calcareous shoreline material there, and the pH values of sediment and bottom water at all 12 field sites sampled in 1983 were close to 7 (see p. 32).

The results of the study suggest that as far as offshore regions of "old" lake bottom are concerned the relatively low rates of Hg methylation in Southern Indian Lake sediment compared with Notigi Reservoir sediment can be attributed not only to the greater proportion of inundated land in Notigi Reservoir but also the greater abundance of clay in Southern Indian Lake - a result of detrital loadings from the Churchill River as well as from shoreline deposits. Most of the suspended detritus is deposited in

Granville Lake and Southern Indian Lake, so that relatively little is transported into Notigi Reservoir. Without more information, it is impossible to compare quantitatively the effects of detrital clay and extent of flooding on the geographical variations in methylation rate. Nevertheless, the strong positive correlation between CH_3Hg^+ production and post-impoundment rise in water level (Fig. 7) suggests that the extent of flooding is the more important factor.

In brief, stimulation of microbial growth by organic nutrients introduced into the aquatic environment from submerged land is the primary cause of the elevated rates of CH_3Hg^+ production in the reservoirs, although the influence of secondary environmental factors must also be taken into account.

Accumulation of Hg by fish.

From the standpoint of human economic interests, the most important species of fish affected by the Churchill-Nelson River diversion are lake whitefish (*Coregonus clupeaformis*), which feed principally on benthic invertebrates, and the piscivorous predators walleye (*Stizostedion vitreum*) and northern pike (*Esox lucius*). In all three species of fish the Hg content of muscle tissue increased as a result of flooding caused by hydroelectric development (Bodaly *et al*., 1984a; McGregor, G.W.G., unpublished data, personal communication), and CH_3Hg^+, as expected, generally predominated over inorganic Hg (Table 2). Despite these broad similarities, however, the geographical and temporal patterns of variation in the Hg content of muscle differ considerably among the three species.

The discussion that follows is based on a synthesis of other workers' estimates of Hg levels in fish (Bodaly *et al*., 1984a; McGregor, G.W.G., unpublished data, personal communication) and the results of the present writer's work on the biogeochemistry of Hg in sediments and water, including selected data from a study carried out before the Canada-Manitoba agreement went into effect (Jackson, Ms. in preparation).

Geographical and temporal variations of Hg levels in fish. Hg concentrations in the muscle tissue of walleye and pike collected from different major regions of the field area in 1980 and 1981 tend to correspond broadly with the extent of flooding and the CH_3Hg^+ generating capability of the bottom sediment, and therefore increase in the order Granville Lake < Southern Indian Lake < Notigi Reservoir lakes (Bodaly *et al.*, 1984a; McGregor, G.W.G., unpublished data, personal communication), although the two species show quite different patterns of geographical variation within the group of lakes comprising Notigi Reservoir (Rat Lake, Notigi Lake, West Mynarski Lake, and Central Mynarski Lake). Whitefish collected from the same lakes in the same years deviated greatly from both walleye and pike: Whereas Hg levels in both walleye and pike were consistently much lower in Southern Indian Lake than in any of the Notigi Reservoir lakes, the Hg content of whitefish was, on average, nearly the same in Southern Indian Lake as in Notigi Lake (and even higher in Southern Indian Lake as far as the 1980 catch is concerned) even though the impact of flooding was proportionately much smaller in Southern Indian Lake than in Notigi Lake. The aberrant results given by whitefish may reflect the influence of dietary sources of Hg - specifically benthic invertebrates, whose Hg levels were found to be lower in the Notigi Reservoir lakes than in Southern Indian Lake, possibly owing to suppression of the bioavailability of inorganic Hg by hydrated iron and manganese oxides and humic matter in the sediments (Jackson, Ms. in preparation). (*Note*: The distribution of Hg in benthic invertebrates of the field area has also been treated by Warwick (1986)).

In addition to these differences in the geographical distribution of Hg accumulations in their tissues, the three species of fish gave distinctly different patterns of variation in Hg content with respect to the time elapsed since impoundment. Although the Hg levels in all three species rose markedly in Southern Indian Lake within two years after the impoundment and diversion of Churchill River water, Hg levels in whitefish peaked rapidly and then declined sharply and progressively, reentering the range of pre-impoundment values by 1982, whereas Hg levels in pike rose continuously and were still rising by 1982, while the Hg content of walleye levelled off well above the baseline values and had not shown any significant change by 1982 (Bodaly et al., 1984a). Furthermore, Hg concentrations were consistently much higher in walleye and pike than in whitefish. Thus, both walleye and pike are efficient accumulators of Hg - CH_3Hg^+ in particular - as expected from the lipophilic character of CH_3Hg^+ and the strong tendency of CH_3Hg^+ to be bound by sulfhydryl groups of proteins, although the two species obviously differ to some extent in their Hg accumulation kinetics. On the other hand, whitefish evidently expel Hg rapidly, even though most of their Hg, too, is in the form of CH_3Hg^+. Further research is needed to explain the fact that whitefish have a far more weakly developed tendency to accumulate Hg than do walleye and pike, but certain plausible working hypotheses may be considered. One possibility is that CH_3Hg^+ taken up by whitefish is excreted rapidly by means of a special biochemical mechanism such as the bonding of CH_3Hg^+ by a "carrier" molecule (perhaps a protein or peptide) to form a hydrophilic, and therefore readily excreted, complex (cf. Norseth and Clarkson, 1971;

Kuohtula and Miettinen, 1975; Farris et al, 1977), or biochemical demethylation of CH_3Hg^+ followed by excretion of the inorganic Hg thereby produced (Norseth and Clarkson, 1970 and 1971; Burrows and Krenkel, 1973). Diet could be an important factor as well. Walleye and pike feed on other fish, which are undoubtedly enriched in CH_3Hg^+ owing to biomagnification during transfer of the CH_3Hg^+ up the food chain, but whitefish feed on benthic invertebrates, which are lower in the food chain and, in any case, commonly have lower CH_3Hg^+ concentrations in proportion to their inorganic Hg concentrations than is usual among fish (Huckabee et al., 1979; Windom and Kendall, 1979); efficient elimination of this inorganic Hg may account, at least in part, for the relatively low total Hg levels in whitefish. Note that in some sampling regions of Southern Indian Lake (specifically the more northerly ones) the proportion of CH_3Hg^+ in the total Hg burden of muscle and liver tissue was lower, on average - and in all sampling regions ranged to much lower minimum values - in whitefish than in walleye and pike (Table 2), suggesting appreciable intake of inorganic Hg, probably from dietary sources. Moreover, the ratio of the total or methyl Hg content of liver to the total or methyl Hg content of muscle was consistently higher in whitefish than in walleye or pike in Southern Indian Lake (Table 2), suggesting that the rates of total and methyl Hg excretion are highest in whitefish (Jernelöv and Lann, 1971). This gives further support to the hypothesis that whitefish eliminate CH_3Hg^+ rapidly in the form of water-soluble complexes.

In the next section an attempt is made to relate the biogeochemistry of Hg in bottom sediments to Hg concentrations in fish. Inasmuch as the biogeochemical data were particularly successful in predicting and explaining variations in the Hg content of walleye, we will deal principally with this species. Effects of different biogeochemical phenomena on the accumulation of Hg by a wider range of aquatic organisms are treated elsewhere (Jackson, Ms. in preparation).

Availability of Hg to walleye. In general, fish preferentially accumulate Hg in the form of CH_3Hg^+, which is synthesized from inorganic Hg(II) by free-living microbes. Most of the CH_3Hg^+ in aquatic environments is thought to be synthesized in surficial bottom sediments, where microbial activity and Hg concentrations are usually greatest, and the CH_3Hg^+ subsequently diffuses out into the water column, where it is taken up by fish, both by direct absorption from water through the gill membrane and from food. Thus, we would expect the kinetics of Hg uptake by fish to depend largely on the concentration of CH_3Hg^+ in the water column. The CH_3Hg^+ content of the water, in turn, is a function of the concentration of biochemically available inorganic Hg(II) in the bottom sediment and the rate at which microbes in the sediment are able to transform this available inorganic Hg into CH_3Hg^+. The available inorganic Hg pool is only a fraction of the total inorganic Hg, the proportion of available to unavailable forms of the metal being a function of environmental factors such as the abundance of sulfides and other Hg-binding agents that render Hg unavailable. The rate of CH_3Hg^+ production, in turn, depends not only on the supply of available inorganic Hg(II) but also on different environmental factors such as nutrient levels, temperature, ambient pH, and oxidation-reduction conditions, which control microbial activity. In addition, of course, there are other factors to be considered, such as food chain effects and the innate characteristics of the fish species. Thus, the Hg content of a fish depends on a multitude of interrelated variables.

Despite the daunting complexity of the biogeochemical cycle of Hg, the writer found empirically that in the lakes of the field area the Hg content of the muscle tissue of walleye (but not pike or whitefish) varies as a simple function of a single easily measured compound variable combining a factor representing the abundance of available inorganic Hg in the bottom sediments and a factor representing the rate at which the sediment converts this Hg into CH_3Hg^+. Thus, the mean Hg concentrations in the muscle tissue of walleye from the lakes sampled gave a very strong positive correlation with the experimentally determined "methylating capability" of the sediment multiplied by the abundance of "available inorganic Hg" as represented by either the $CaCl_2$-extractable Hg concentration or the Hg/S^{2-} or Hg/Se ratio of the sediment (Fig. 25, A-C). Not surprisingly, the concentrations of CH_3Hg^+ in bottom sediments and overlying surface water (represented by samples collected in June, 1981) also correlate significantly with this index of CH_3Hg^+ production in the sediment (Fig. 26, A and B) (See also Warwick, 1986). Note that the size of the fish (as measured by fork length or by weight) does not correlate significantly with the Hg content of the fish, and therefore did not have to be taken into account. In any case, Hg content normalized with respect to size gave virtually the same results as Hg content alone, showing that normalization was superfluous.

These observations support the reasonable assumption that the major pathway for accumulation of Hg by walleye consists of CH_3Hg^+ production in bottom sediments , diffusion of this CH_3Hg^+ into the water column, and highly efficient uptake and accumulation of the CH_3Hg^+ by the fish.

At this point, it is necessary to digress briefly and deal with a question of statistical methodology that arises in connection with the plots shown in Figures 25 and 26. Since these plots and the accompanying statistical data could be criticized on the grounds that regression analysis employing compound variables may give ambiguous, misleading, or even spurious results, it should be pointed out that the writer examined not only the relationships illustrated in the diagrams but also the relationship between the Hg content of the fish and each of the individual variables of which the compound variables are composed (i. e. the methylation capability and total Hg, $CaCl_2$-extractable Hg, sulfide, and selenium concentrations of the sediment). Not only did the compound variables give much stronger correlations which were highly signficant ($P < 0.001$ or $P > 0.001, < 0.01$), but most of the individual variables, when tested separately, gave decidedly insignificant correlations ($P > 0.1$). These observations argue persuasively for the validity of the compound parameters as indicators of the combined effect of the principal factors on which the processes of CH_3Hg^+ production in sediments and CH_3Hg^+ accumulation by walleye depend. It is the combination of factors which is of decisive importance in regulating these processes, not any one of the component factors by itself. Thus, we may conclude that the correlations shown in Figures 25 and 26, and the theory on which they are based, are valid and are useful for quantitative (or semi-quantitative) comparison of different lakes in the field area and assessment of the effects of hydroelectric development on the biogeochemical cycling of Hg in those lakes.

To summarize, the results presented in Figures 25 and 26 lead to the following conclusions: (1) The rate of Hg (specifically, CH_3Hg^+) accumulation by walleye muscle is a function of the CH_3Hg^+ concentration in the water. (2) The CH_3Hg^+ concentration in the water is, in turn, controlled by the rates of CH_3Hg^+ production by microbes in bottom sediments and subsequent release of the CH_3Hg^+ into the water column. (3) The rate of CH_3Hg^+ production in bottom sediment depends on two variables, (a) the concentration of readily solubilized, "available" inorganic Hg and (b) the degree of heterotrophic microbial activity, which determines the rate at which the available inorganic Hg is methylated. As demonstrated in this report and elsewhere (Jackson, Ms. in preparation), flooding caused by the Churchill-Nelson River diversion has increased both of these essential factors, as well as the total abundance of Hg, for they all show a general tendency to be greatest where the environmental impact of flooding has been most pronounced and widespread, although effects of local environmental influences complicate the picture and cause deviations from this broad pattern of variation. Presumably the available and unavailable forms of Hg are both largely derived from recently submerged plant remains and and soil organic matter, and of course these same organic materials are also primarily responsible for stimulating the activities of methylating microbes. As shown in Figures 25 and 26, the severity of the Hg problem caused by the effects of flooding increases in the following order: Granville Lake < South Bay (Southern Indian Lake) < Rat Lake < West/Central Mynarski Lakes < Notigi Lake.

We have examined in detail the environmental conditions that control the activities of Hg methylating and demethylating microbes in sediments, and have considered evidence pertaining to the origin of the Hg. There remains, however, an important and fundamental question: What are the environmental factors which determine the proportion of the total sediment-bound inorganic Hg pool that is available for methylation? A possible answer is provided by Figure 28, in which the abundance of the $CaCl_2$-extractable Hg fraction, expressed as percentage of the total Hg, is plotted against the manganese/iron ratio of the $NH_2OH \cdot HCl$-extractable hydrated oxide fraction of the sediment. The diagram reveals a highly significant positive correlation, and the samples from "main-stem" regions, where the water is rapidly mixed, have higher proportions of $CaCl_2$-extractable Hg and higher manganese/iron ratios than do the samples from the "backwater" sites, where the water is comparatively stagnant. This relationship is interpreted as signifying that vigorous aeration (which is typical of main-stem environments) enhances the solubility and availability of the inorganic Hg - probably by promoting oxidation of sulfides and organics - while fostering a high manganese/iron ratio owing to precipitation of manganese as MnO_2 or MnOOH. (Note that manganese nodules were found at two of the main-stem sites but at none of the backwater sites).

Thus, aeration appears to have two conflicting effects as far as the Hg problem is concerned: Oxidizing conditions tend to lower the rate of methylation, as demonstrated by the experimental results presented above, but they also increase the mobility and availability of inorganic Hg. The net effect of aeration on the biological uptake of Hg under field conditions would be difficult to judge without more research. Figure 28 indicates a relatively large disparity between the inorganic Hg availability data for sediments from the main-stem west basin and the backwater east basin of Notigi Lake, but Figure 26A shows only a slight difference between the CH_3Hg^+ concentrations found in the same sediment samples. Possibly, therefore, the two opposing effects of oxidation-reduction conditions very nearly offset one another.

Finally, there is one additional question that demands an answer: Why do the Hg levels in pike not show the same patterns of variation as the Hg levels in walleye? We have already discussed possible reasons for the fact that whitefish are highly anomalous with respect to walleye and pike; but considering that walleye and pike are both situated at the upper end of the food chain and have similar feeding habits (both being piscivorous predators which eat a wide variety of prey (Scott and Crossman, 1973)), and considering that they both accumulate CH_3Hg^+ efficiently and show no tendency to eliminate Hg rapidly, it is not immediately clear why strong correlations between the Hg content of fish muscle and CH_3Hg^+ concentrations and production rates in the environment were found in walleye but not in pike. Further study is needed to explain this, but differences in habitat preference (and, perhaps, a bias in the selection of sites for sampling sediments and water) may provide at least a partial explanation. Walleye mainly inhabit open water, whereas pike prefer to lurk in shallow, weedy environments. Since most of the sediments and water collected for this study came from the deeper, offshore regions of the lakes, the geochemical data related to the production and abundance of CH_3Hg^+ may be more representative of the environment of walleye than the environment most favoured by pike. Jackson (Ms. in preparation) reported that Hg levels in pike correlate strongly with NaOH-extractable iron in the bottom sediments, suggesting an influence of nearshore environments rich in humic-iron complexes derived from the surrounding land and from recently submerged soil. Conceivably the higher metabolic rate and less sedentary habits of walleye also have something to do with the question. Walleye, owing to their greater tendency to move from place to place, may have CH_3Hg^+ body burdens that more accurately reflect the average CH_3Hg^+ concentrations of the water and sediments and the average CH_3Hg^+ production rates of the sediments.

ACKNOWLEDGMENTS

The field work was carried out by J. Tisdale assisted by D. Steinke and E. DuVal. Chemical analyses of field samples were performed by K. Supeene, J. Tisdale, and technicians of the Chemical Research and Analytical Services section of the Freshwater Institute. Most of the experiments, including associated analytical work, were performed by A. Adams, and an additional experimental project was performed by C. Ford. Throughout the experimental work, technical assistance and guidance were provided as needed by K. Supeene. X-ray diffraction and grain-size analyses of shoreline clay were performed by Dr. W.M. Last (Department of Earth Sciences, University of Manitoba), and grain-size analyses of bottom sediments were done by E. DuVal. I gratefully acknowledge the valuable help of colleagues at the Freshwater Institute, in particular the following: (1) R.E. Hecky for information on and advice which assisted both the field work and the laboratory studies, and for assistance and cooperation in connection with sample collection; (2) A. Furutani for permission to use his gas chromatograph for CO_2 and CH_4 analyses and for advice concerning experimental technique; (3) R.A. Bodaly, J.F. Klaverkamp, and M. Giles for information on fish; and (4) G.W.G. McGregor for kindly and generously allowing me to use his unpublished data on Hg in fish populations of Southern Indian Lake.

I also thank R.A. Bodaly, L. McKerness, E.D. Ongley, P. Ramlal, and R. Webster for reading the manuscript and offering useful critical comments. Funding for the research was provided through the agency of the Steering Committee for the Canada-Manitoba Mercury Agreement, whose support is gratefully acknowledged.

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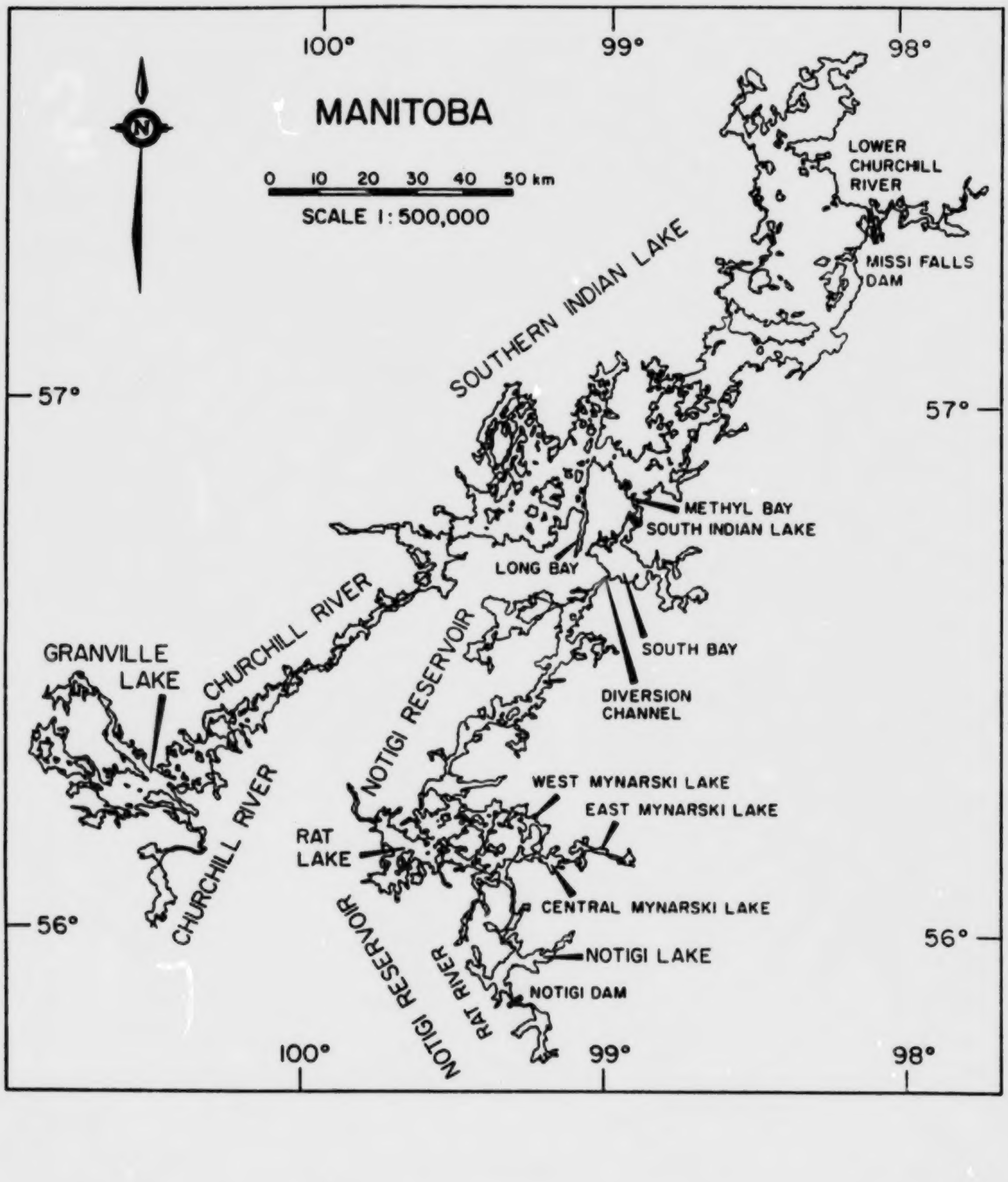
Table 1. The organic carbon (C) and nitrogen (N) concentrations of the bottom sediments and the quantities of CH_3Hg^+ generated on incubation of sediment samples for 10 days under N_2 in the presence of $HgCl_2$ solution without nutrient enrichment. All data are expressed as amount per unit dry weight of sediment.

Sampling site	Organic C (mg/g)	N (mg/g)	CH_3Hg^+ (ng generated per g)
Granville Lake	25.3	3.4	4.96
Southern Indian Lake			
South Bay	16.0	1.2	5.11
Long Bay (central basin)	15.0	1.6	9.25
Long Bay (shallow cove)	31.0	3.4	17.1
Methyl Bay (offshore)	31.0	3.1	5.58
Methyl Bay (flooded forest)	161	8.2	741.9
Notigi Reservoir			
Rat Lake	21.0	2.0	11.0
Notigi Lake (west)	34.5	4.0	23.8
Notigi Lake (east)	39.0	4.5	12.8
West Mynarski Lake	104	13.2	43.3
Central Mynarski Lake	77.5	9.8	20.3
East Mynarski Lake	108	13.4	91.1

Table 2. Ratios computed from total Hg and CH_3Hg^+ concentrations in muscle and liver of walleye, pike, and whitefish collected from four regions of Southern Indian Lake (Fig. 1) in 1980 (courtesy of G.W.G. McGregor, unpublished data, personal communication). All analytical data were computed on the basis of weight of total Hg or CH_3Hg^+ per unit fresh weight of tissue. Explanation of abbreviations: N = no. of samples; X = mean.

Sampling region*	Species	CH_3Hg^+/Total Hg						Liver Hg/Muscle Hg			Liver CH_3Hg^+/Muscle CH_3Hg^+		
		muscle			liver								
	of fish	N	X	range	N	X	range	N	X	range	N	X	range
South Bay	walleye	19	0.825	0.702-0.962	18	0.890	0.667-1.2	18	0.384	0.204-0.516	18	0.418	0.196-0.580
	pike	20	0.867	0.725-1.01	20	0.894	0.600-1.3	20	0.665	0.290-1.2	19	0.697	0.319-1.4
	whitefish	20	0.836	0.462-1.25	19	0.858	0.556-1.25	26	1.52	1-2.4	19	1.64	1-3.3
Vicinity of the village of South Indian Lake	walleye	23	0.872	0.735-1.0	22	0.869	0.400-1.3	25	0.356	0.282-1.00	22	0.396	0.143-0.833
	pike	21	0.824	0.644-0.981	20	0.825	0.491-1.2	20	0.555	0.275-0.851	20	0.541	0.340-0.724
	whitefish	21	0.923	0.375-1.5	21	0.828	0.143-1.0	27	1.58	0.875-2.5	20	1.50	0.333-2.1
A region NE of the village of South Indian Lake and SW of Missi Falls dam	walleye	14	0.806	0.685-0.932	14	0.897	0.611-1.1	14	0.449	0.197-0.909	14	0.500	0.206-1.0
	pike	24	0.832	0.683-0.985	23	0.758	0.295-1.0	24	0.598	0.373-1.1	24	0.544	0.183-0.987
	whitefish	25	0.795	0.333-1.4	23	0.749	0.455-1.1	24	1.54	0.60-2.9	23	1.45	0.583-2.57
Vicinity of Missi Falls dam	walleye	4	0.877	0.778-0.964	4	0.996	0.909-1.1	4	0.259	0.244-0.278	4	0.296	0.266-0.354
	pike	20	0.898	0.745-1.0	19	0.900	0.744-1.0	20	0.561	0.404-0.780	19	0.556	0.370-0.790
	whitefish	19	0.781	0.25-1.25	19	0.858	0.524-1.6	19	1.60	0.611-2.4	19	1.88	0.923-6.0

*From top to bottom, these are areas 1,2,3, and 4, in that order, in Figure 1 of McGregor (1980).



100°
99°
98°
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LOWER CHURCHILL RIVER
MISSI FALLS DAM
SOUTHERN INDIAN LAKE
57°
57°
METHYL BAY
SOUTH INDIAN LAKE
LONG BAY
CHURCHILL RIVER
NOTIGI RESERVOIR
SOUTH BAY
GRANVILLE LAKE
DIVERSION CHANNEL
CHURCHILL RIVER
WEST MYNARSKI LAKE
EAST MYNARSKI LAKE
RAT LAKE
CENTRAL MYNARSKI LAKE
56°
56°
NOTIGI RESERVOIR
RAT RIVER
NOTIGI LAKE
NOTIGI DAM
100°
99°
98°

Fig. 1. Map of the field area.

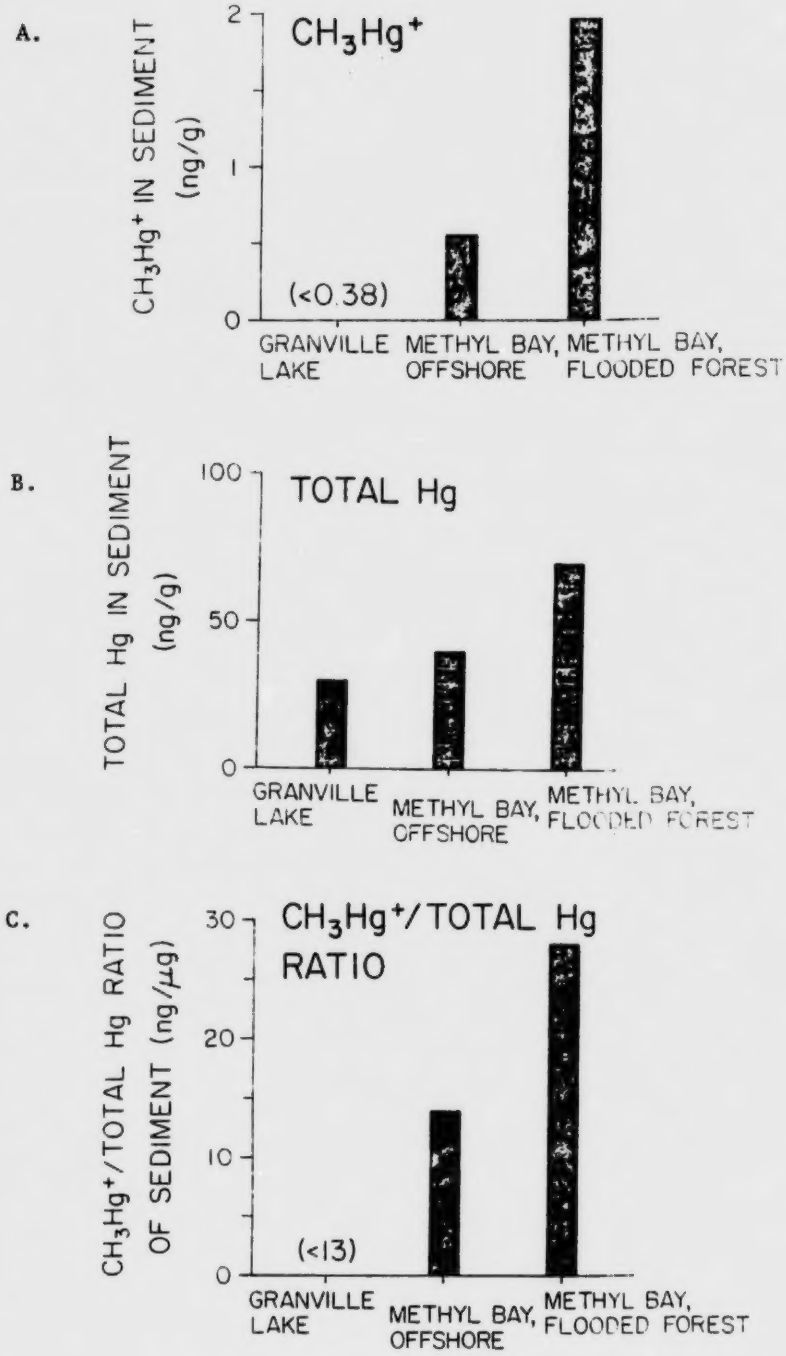

Fig. 2. CH_3Hg^+ and total Hg concentrations, and CH_3Hg^+/total Hg ratios, of sediments collected from Granville Lake and the offshore and flooded forest zones of Methyl Bay (Southern Indian Lake) in August, 1983.

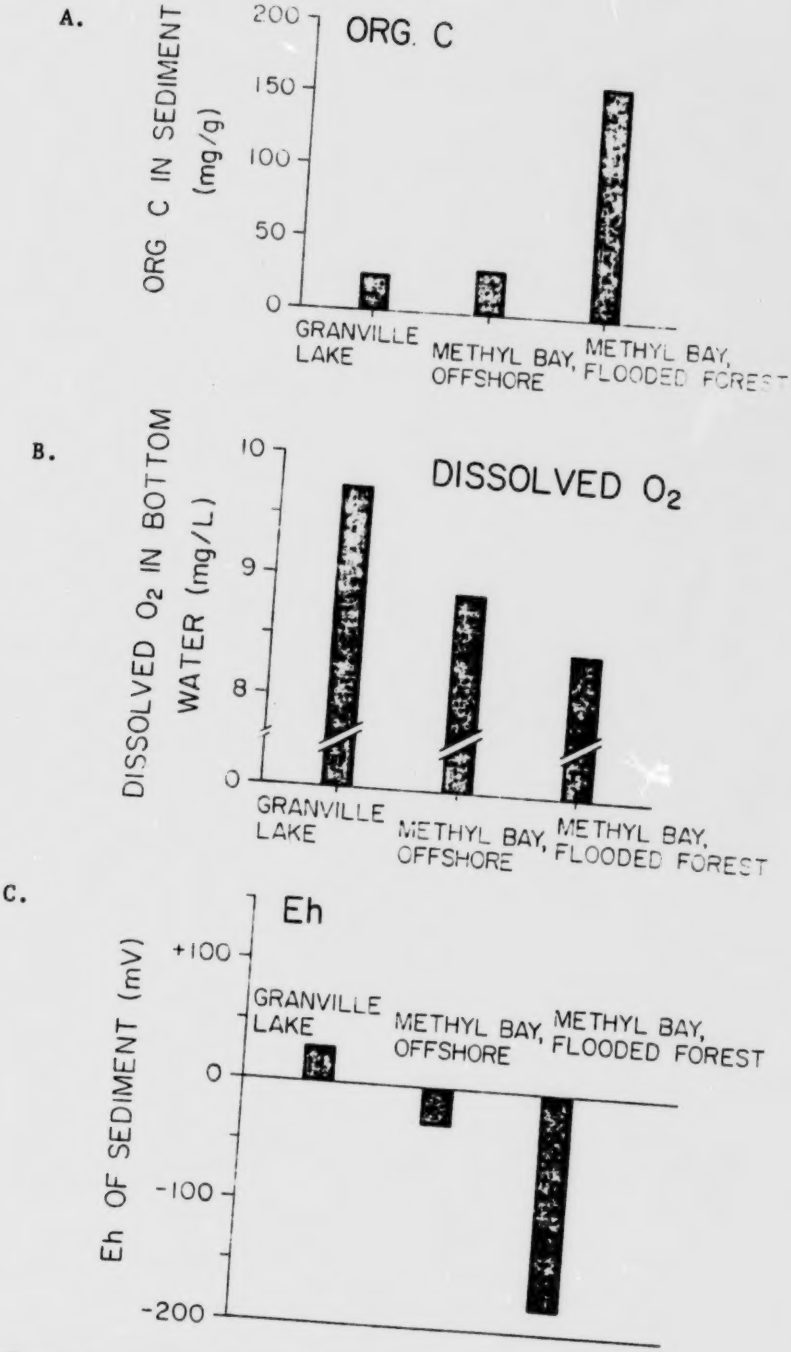

Fig 3. Organic carbon (org. C) concentrations and Eh values of sediments collected from Granville Lake and the offshore and flooded forest zones of Methyl Bay (Southern Indian Lake) in August, 1983, together with dissolved O_2 levels in the overlying water.

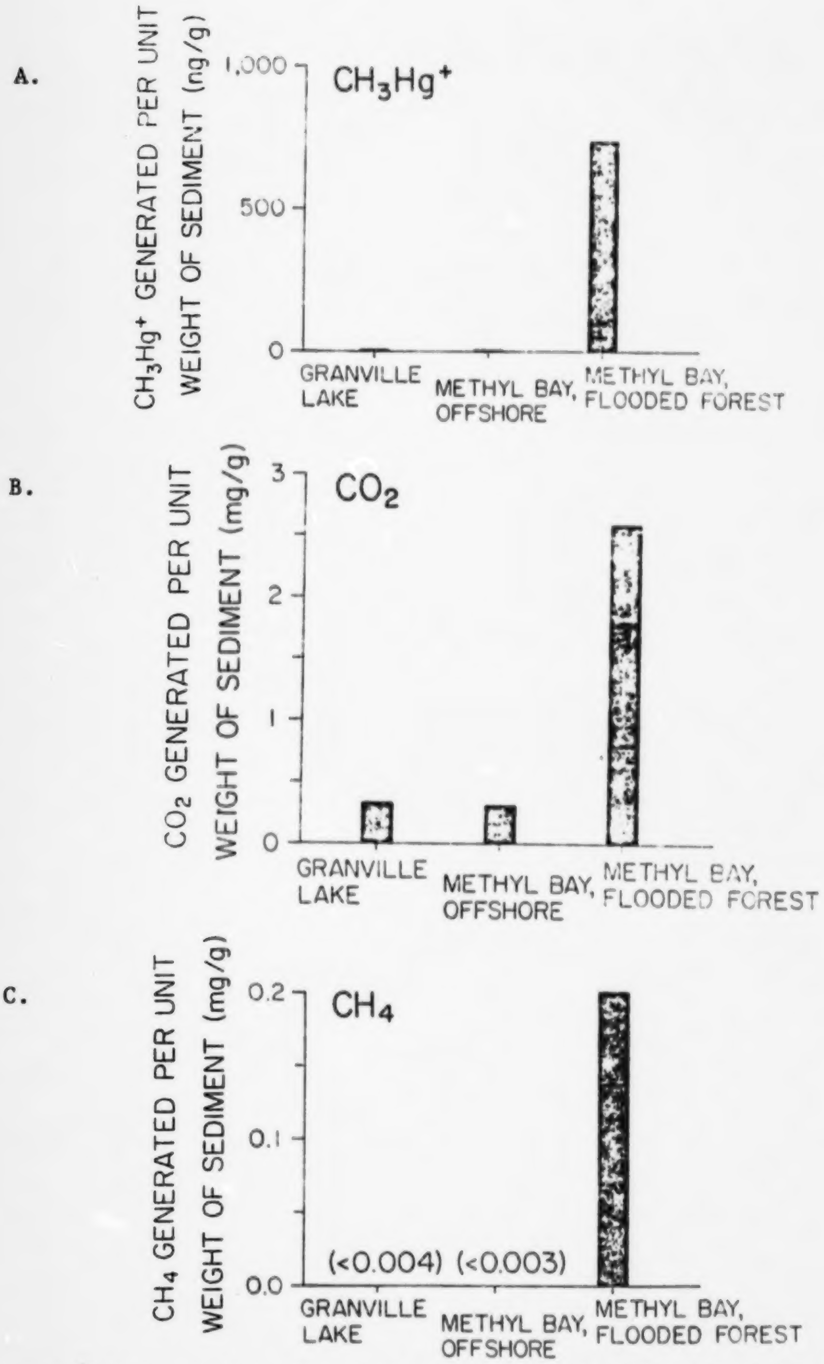

Fig 4. CH_3Hg^+, CO_2 and CH_4 generated by sediments from Granville Lake and the offshore and flooded forest zones of Methyl Bay (Southern Indian Lake) during 10 days' incubation under N_2 in the presence of $HgCl_2$ without nutrient enrichment.

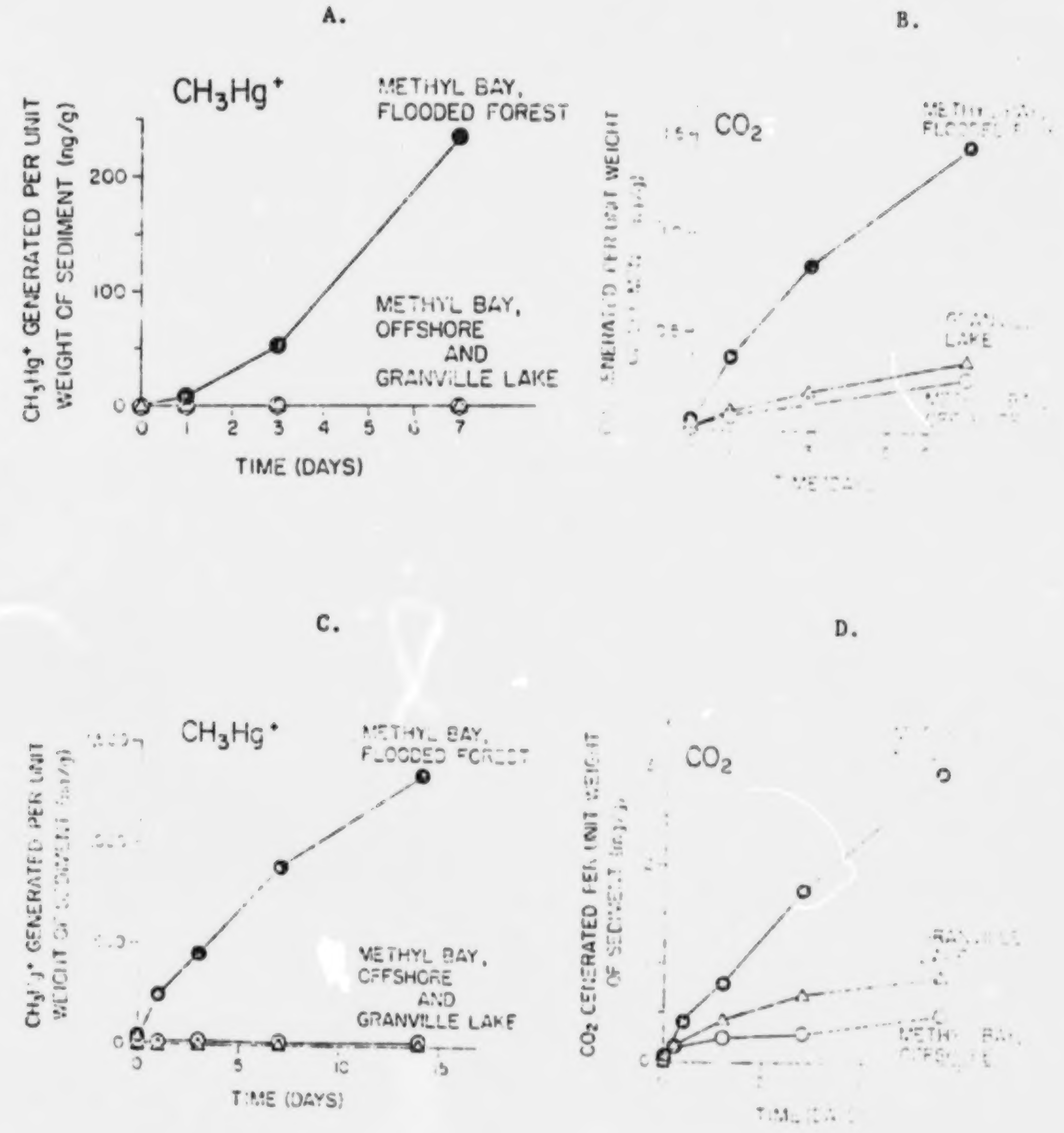

Fig. 5. Variations in the amounts of CH_3Hg^+ and CO_2 generated by sediments from Granville Lake (△) and the offshore zone (○) and flooded forest zone (●) of Methyl Bay (Southern Indian Lake) as functions of incubation time. The sediments, which were collected in August, 1983 (Fig. 5, A & B) and June, 1984 (Fig. 5, C & D), were incubated under N_2 in the presence of $HgCl_2$ without nutrient enrichment.

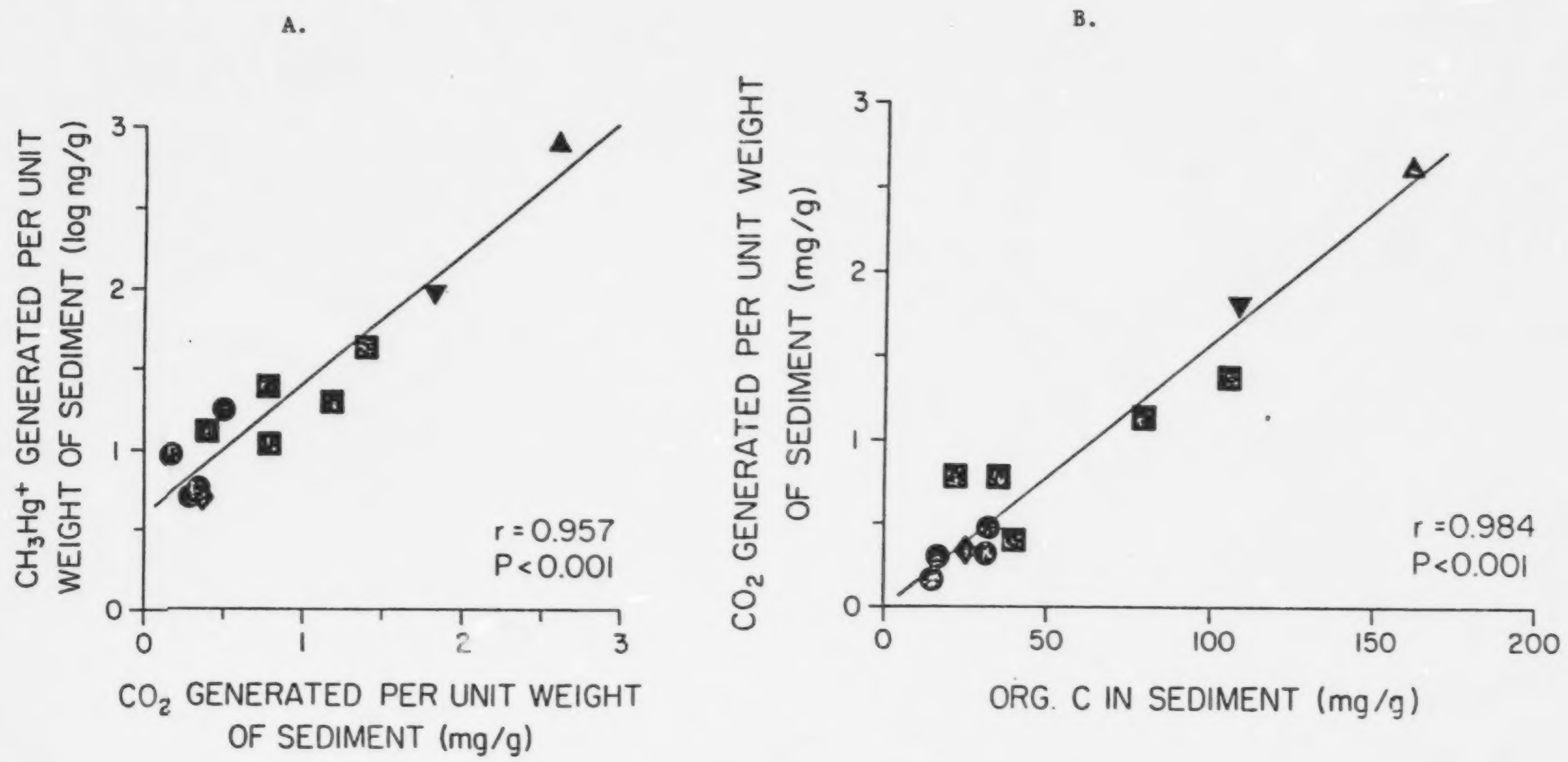

Fig. 6. Relationship between (A) CH_3Hg^+ production and CO_2 production and (B) CO_2 production and organic carbon (org. C) concentration in sediments from different sites. The sediments were incubated for 10 days under N_2 in the presence of $HgCl_2$ without nutrient enrichment. Explanation of symbols: Methyl Bay (Southern Indian Lake) flooded forest site, ▲; other Southern Indian Lake sites, ●; Notigi Reservoir lakes, ■; Granville Lake, ◆; East Mynarski Lake, ▼.

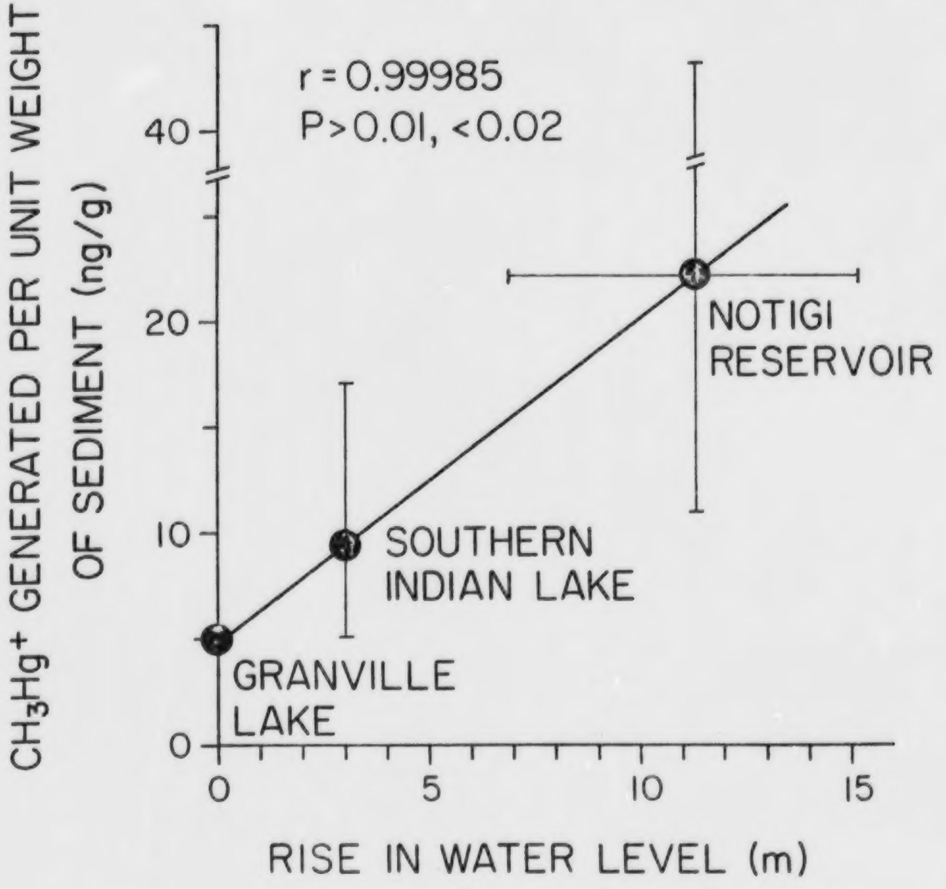

Fig 7. Relationship between CH_3Hg^+ production by various sediments and the impact of flooding as represented by the rise in water level following impoundment. The points on the plot represent mean values. The sediments were incubated for 10 days under N_2 in the presence of $HgCl_2$ without nutrient enrichment.

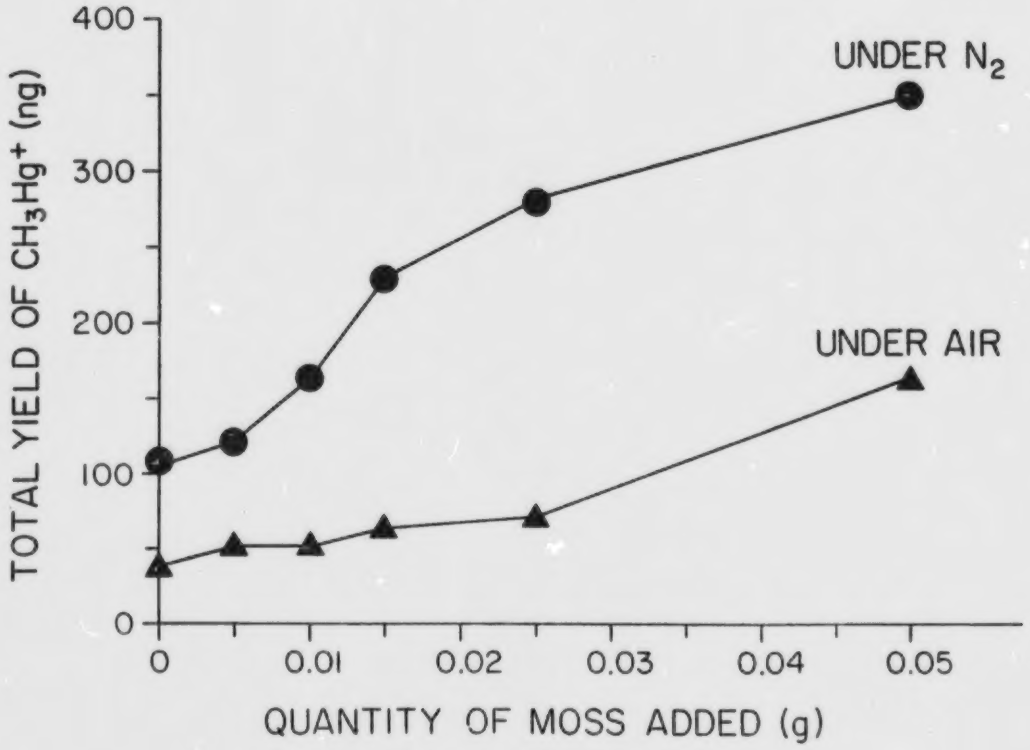

Fig. 8. Effect of nutrient substrate (sphagnum moss) on methylation of $HgCl_2$ by East Mynarski Lake sediment under N_2 (●) and air (▲).

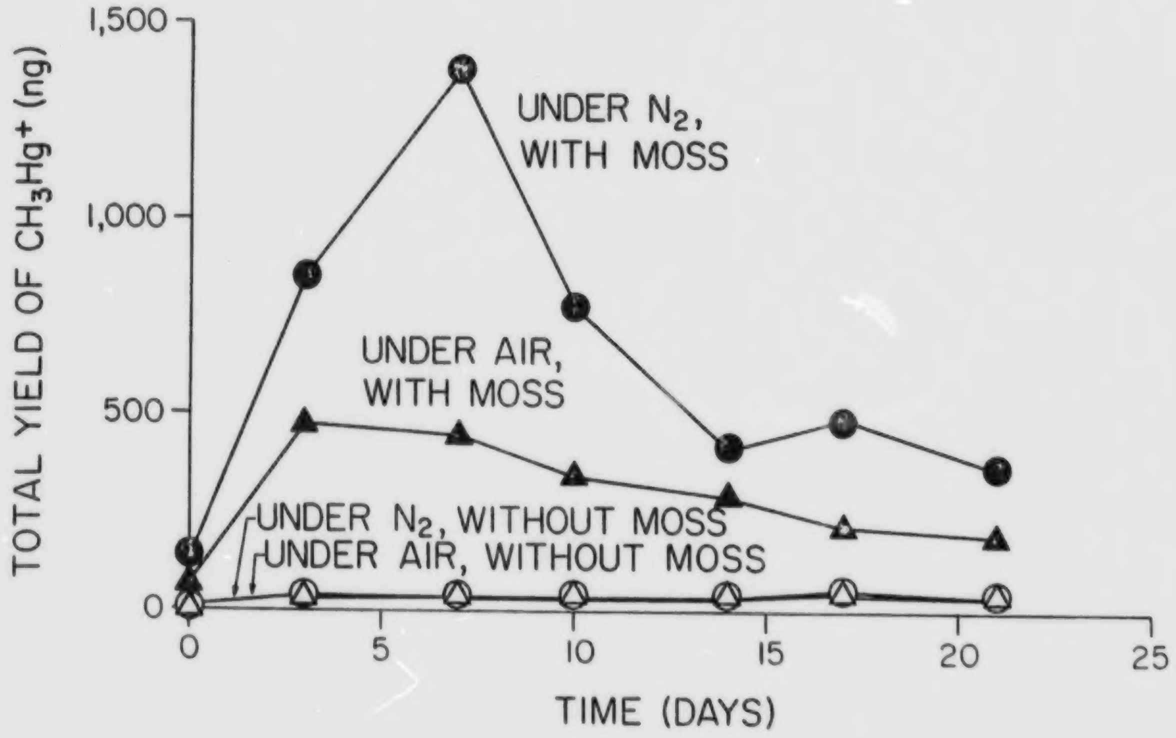

Fig. 9. Production of CH_3Hg^+ from $HgCl_2$ by $CaCO_3$-buffered East Mynarski Lake sediment in the presence and absence of nutrient substrate (moss), under atmospheres of N_2 and air. Explanation of symbols: (1) With moss: under N_2, ● ; under air, ▲ . (2) Without moss: under N_2, ○ ; under air, △ .

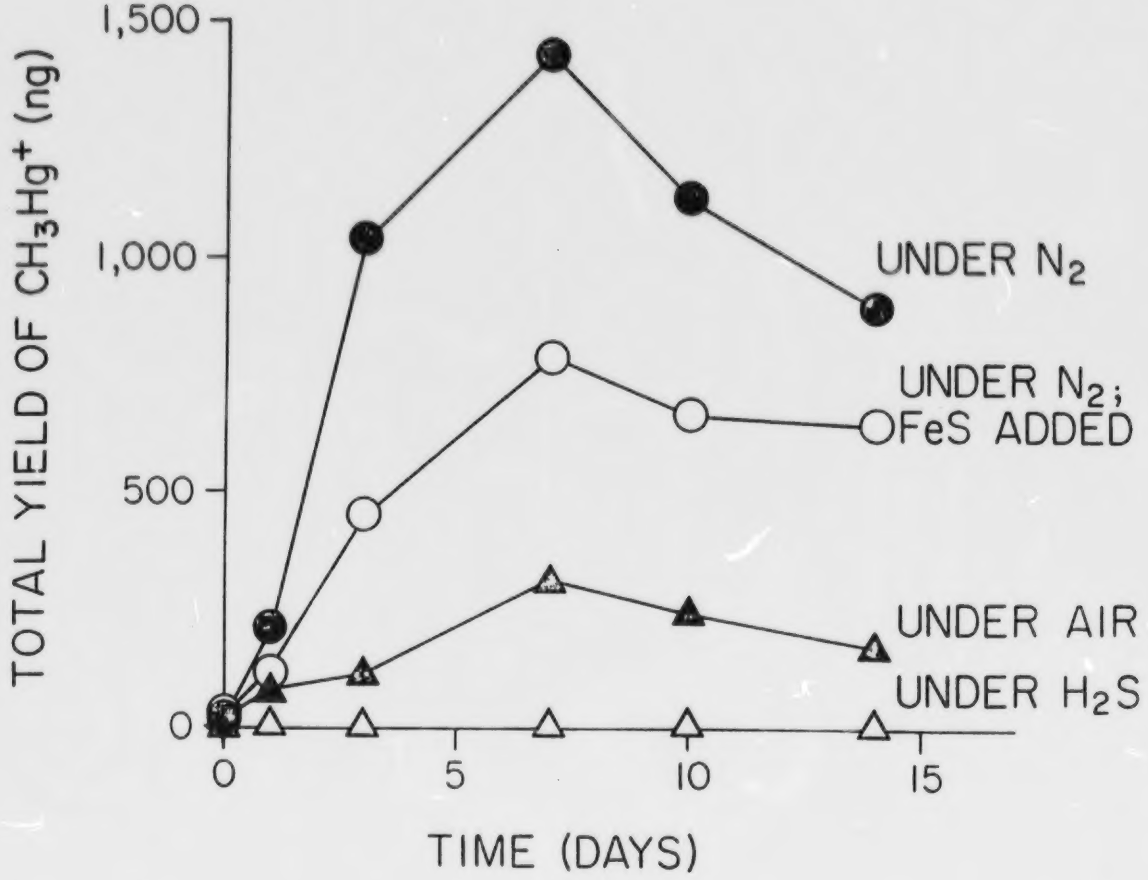

Fig 10. Production of CH_3Hg^+ from $HgCl_2$ by East Mynarski Lake sediment in the presence of moss and $CaCO_3$ under atmospheres of N_2 (●), air (▲), and H_2S (△), and under N_2 with addition of FeS (○).

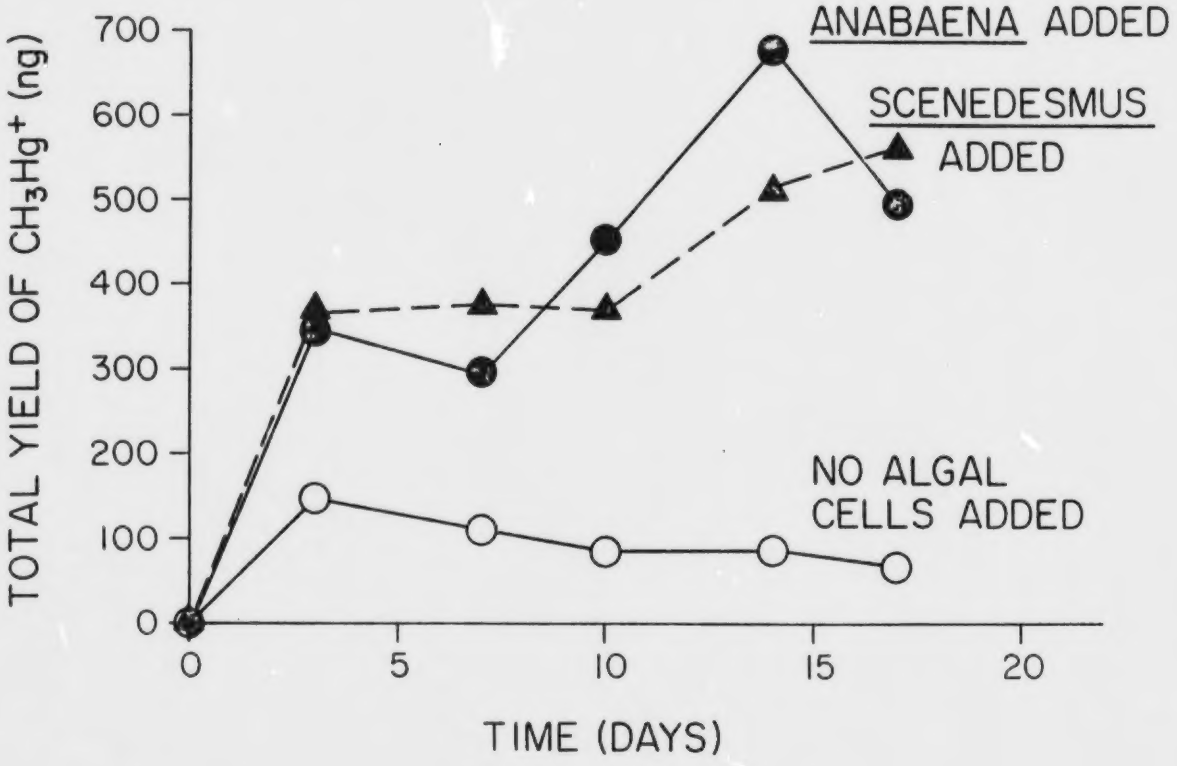

Fig. 11. Effects of nutrient substrates (*Anabaena* and *Scenedesmus* cells) on production of CH_3Hg^+ from $HgCl_2$ by East Mynarski Lake sediment under an atmosphere of N_2. Explanation of symbols: Control, ○ ; *Anabaena* added, ● ; *Scenedesmus* added, ▲ .

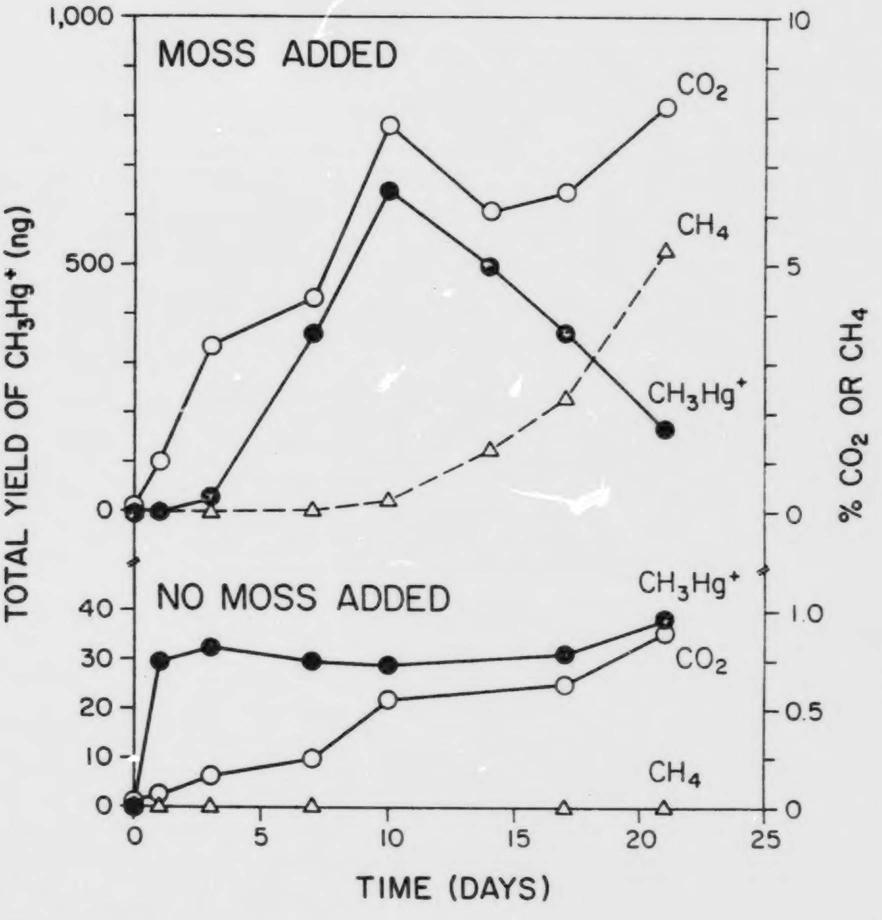

Fig. 12. Production of CH_3Hg^+ (●), CO_2 (○), and CH_4 (△) by Notigi Lake (east basin) sediment under N_2 in the presence of $HgCl_2$ with and without the addition of nutrient substrate (moss).

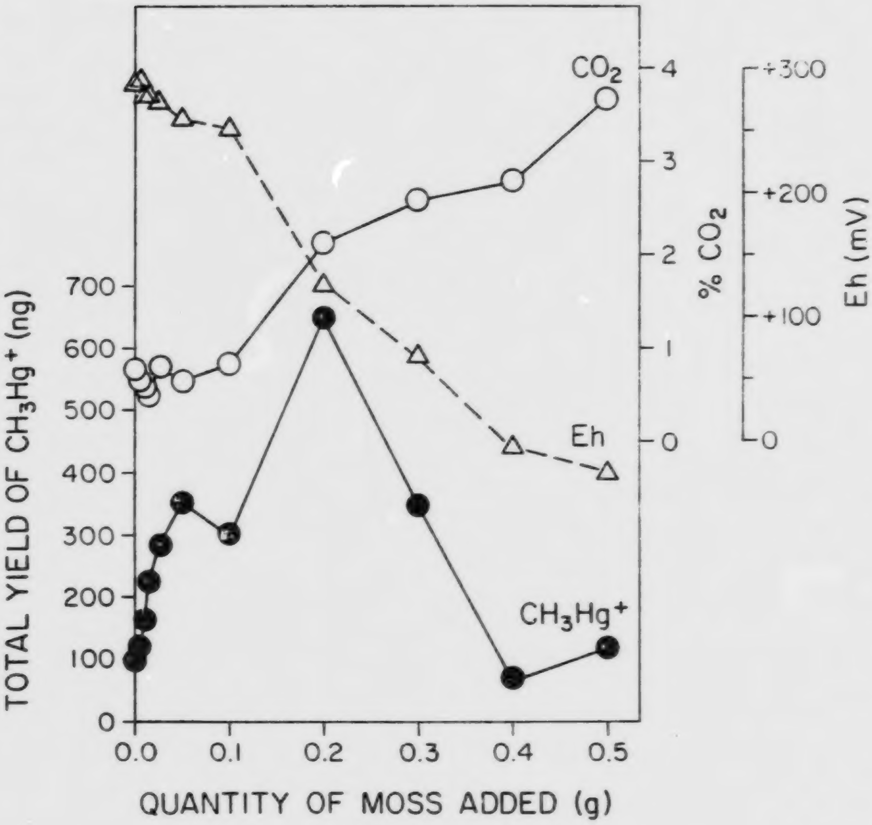

Fig. 13. Effects of varying quantities of nutrient substrate (moss) on CH_3Hg^+ (●) and CO_2 (○) production and Eh (△) in a methylation experiment employing East Mynarski Lake sediment incubated for 7.6 days under N_2 in the presence of $HgCl_2$.

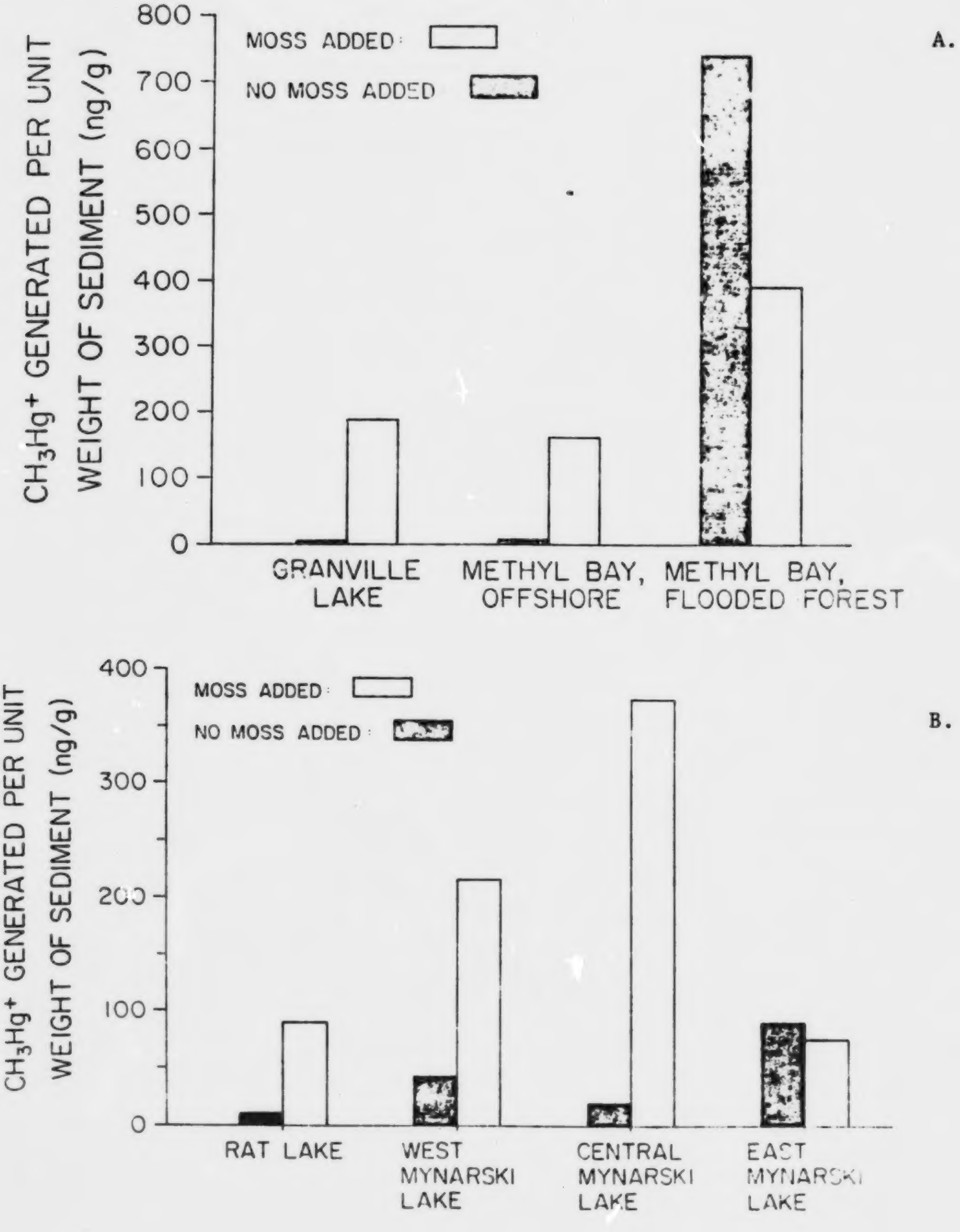

Fig. 14. CH_3Hg^+ produced from $HgCl_2$ by various sediments on incubation for 10 days under N_2 in the presence and absence of nutrient substrate (moss).

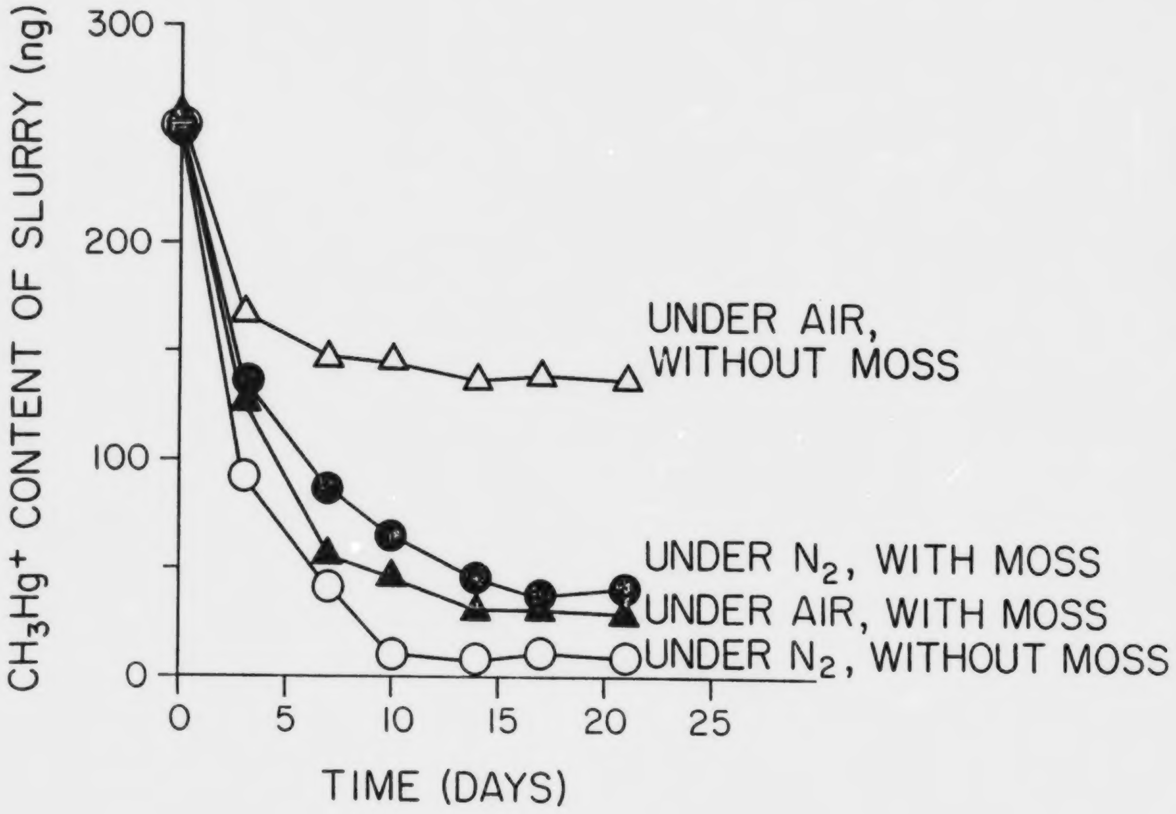

Fig. 15. Demethylation of CH_3Hg^+ acetate by $CaCO_3$-buffered East Mynarski Lake sediment under N_2 and air in the presence and absence of nutrient substrate (moss). Explanation of symbols: (1) With moss: under N_2, ●; under air, ▲. (2) Without moss: under N_2, ○; under air, △.

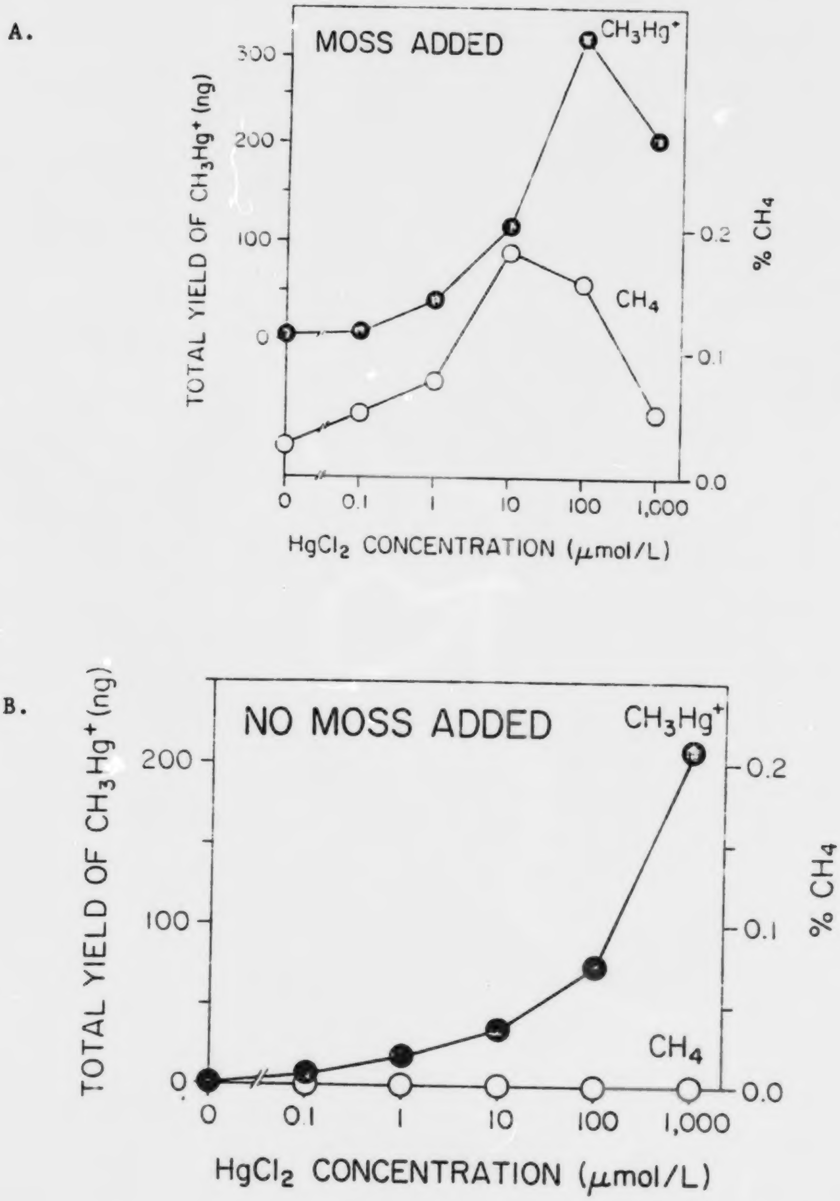

Fig. 16. Production of CH_3Hg^+ (●) and CH_4 (○) by East Notigi Lake sediment at varying initial concentrations of $HgCl_2$ on incubation for 8 days under N_2 with and without added nutrient substrate (moss).

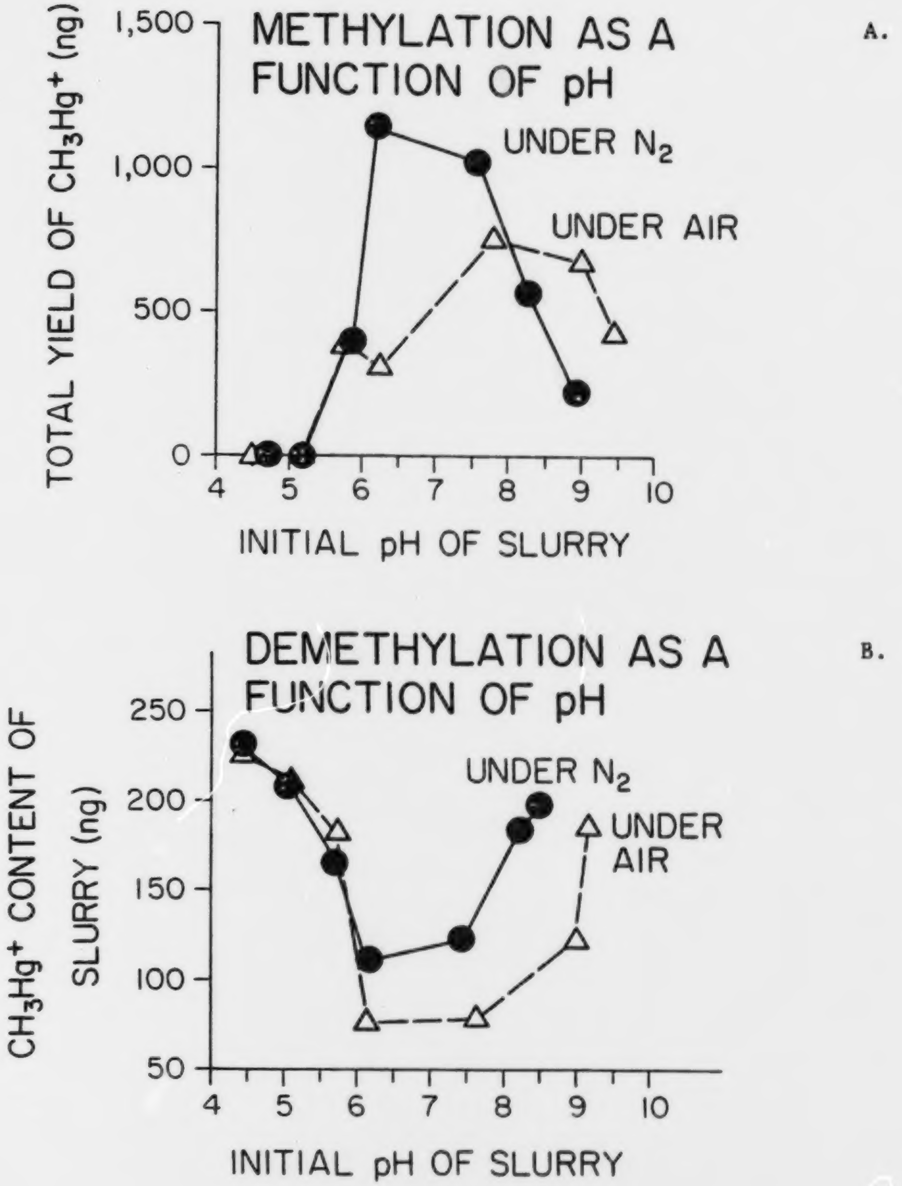

Fig. 17. Effects of pH on methylation and demethylation of Hg by East Mynarski Lake sediment enriched with nutrient substrate (moss) and incubated for 7 days in the presence of $HgCl_2$ or CH_3Hg^+ acetate under atmospheres of N_2 (●) and air (△). The buffers used, and their approximate initial pH values, were as follows: pH 4 & 5 - potassium hydrogen phthalate; pH 6 - KH_2PO_4; pH 7, 8, & 9 - tris(hydroxymethyl)-aminomethane; pH 10 - borax.

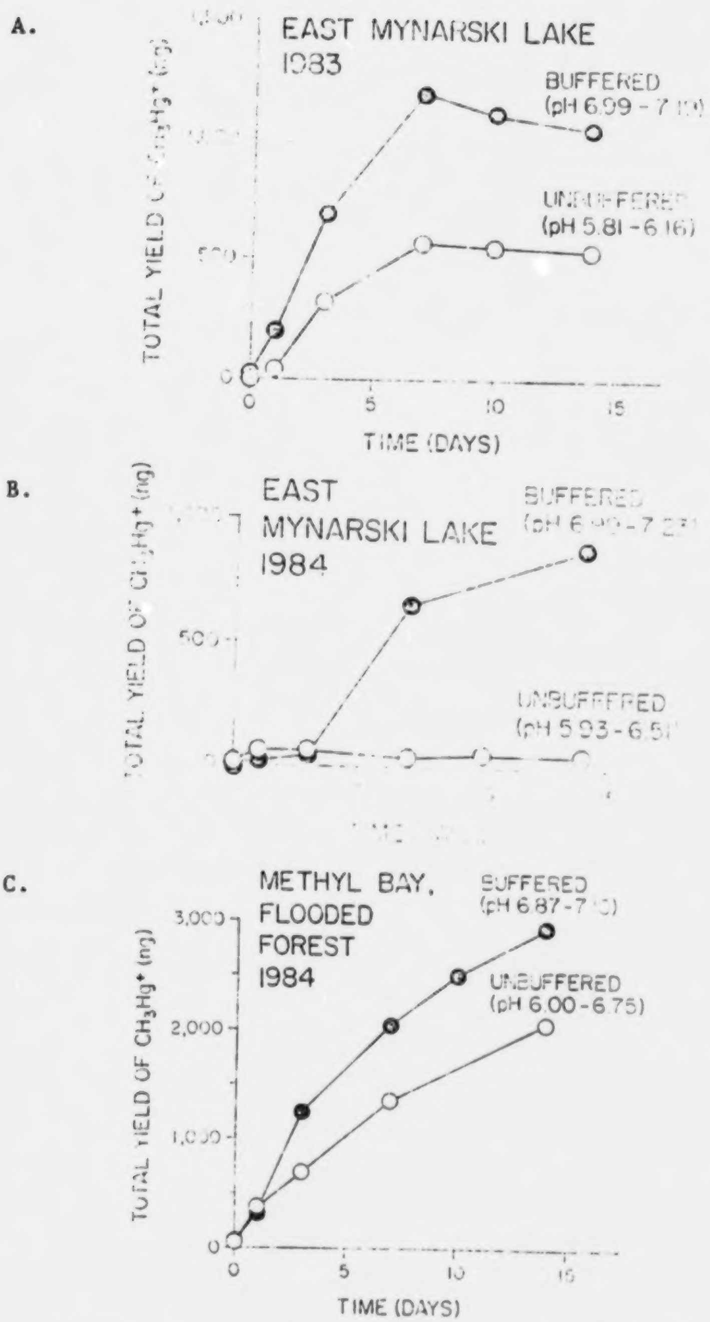

Fig. 18. Effect of buffering with $CaCO_3$ on the production of CH_3Hg^+ by sediments from the flooded forest zone of Methyl Bay (Southern Indian Lake) and East Mynarski Lake. The sediments were incubated under N_2 in the presence of $HgCl_2$; East Mynarski Lake sediment collected in 1983 was enriched with nutrient substrate (moss), but the other sediments were not. Explanation of symbols: $CaCO_3$ added, ●; $CaCO_3$ not added, ○.

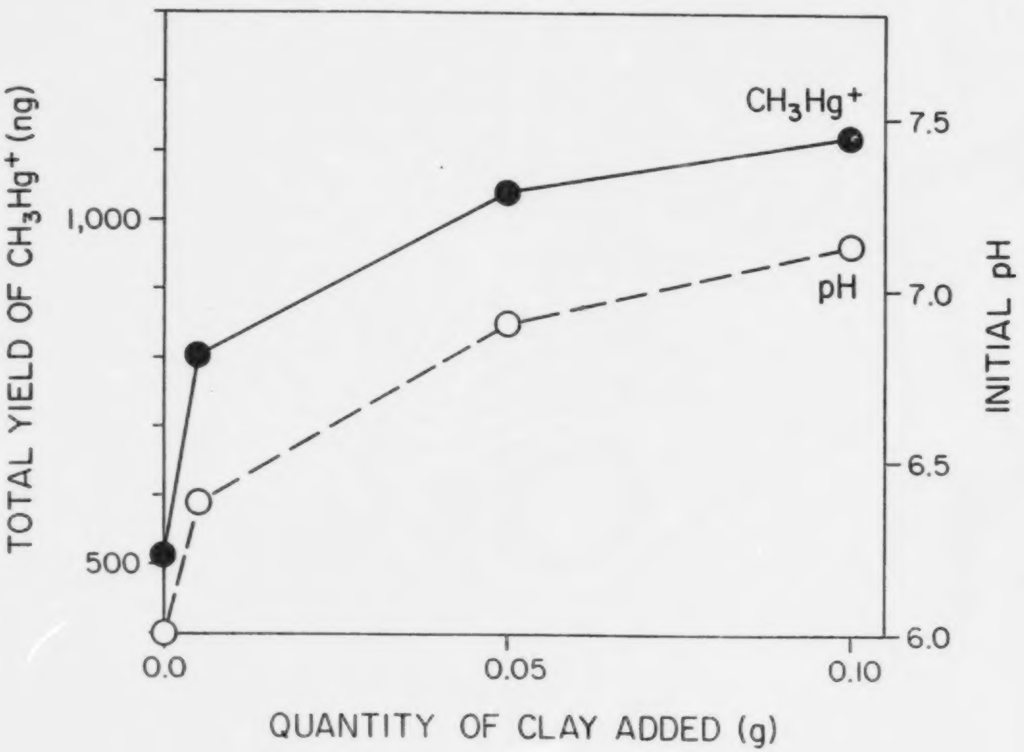

Fig. 19. Effect of shoreline clay on CH_3Hg^+ production (●) and pH (○) in slurries containing East Mynarski Lake sediment, $HgCl_2$, and nutrient substrate (moss). The slurries were incubated for 7 days under N_2.

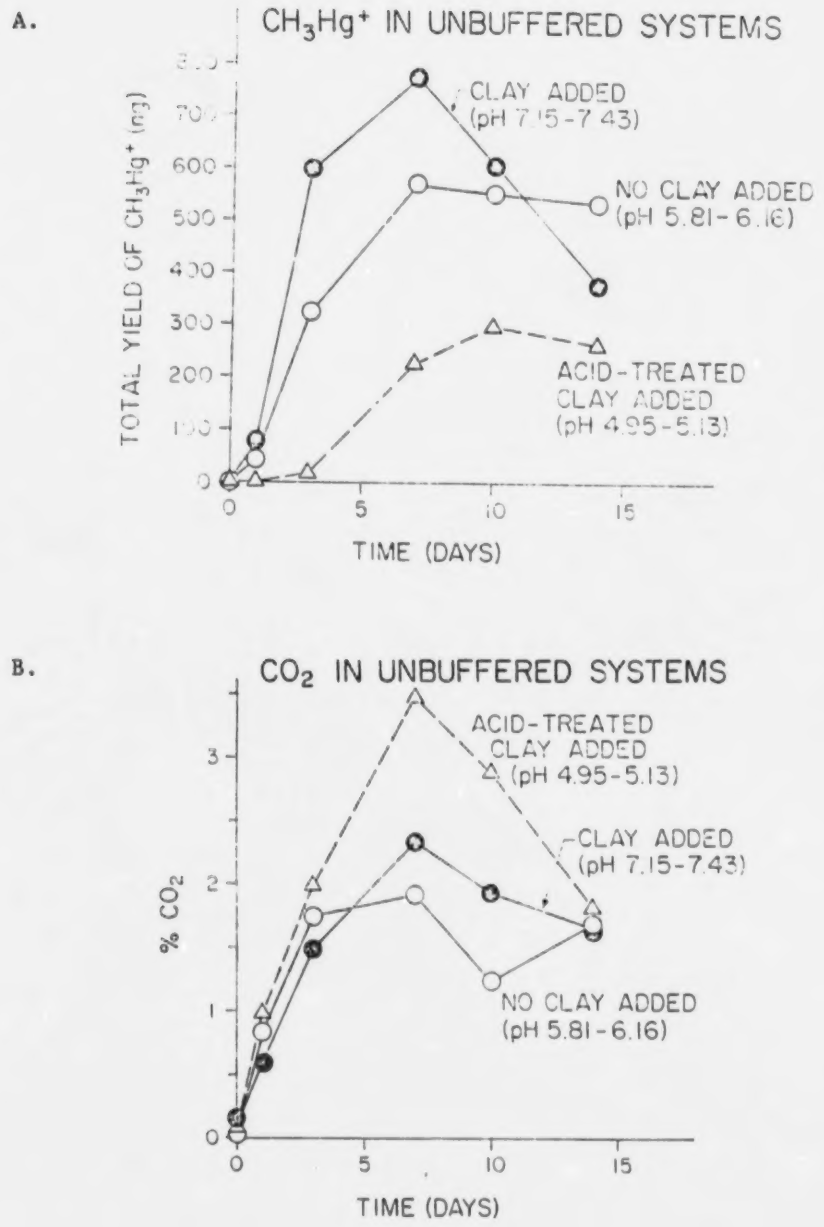

Fig 20. Effects of untreated and acid-leached shoreline clay on production and destruction of CH_3Hg^+ and CO_2 by East Mynarski Lake sediment incubated under N_2 in the presence of $HgCl_2$ and nutrient substrate (moss) but without added buffer. Explanation of symbols: Control, ○ ; untreated clay added, ● ; acid-treated clay added, △ .

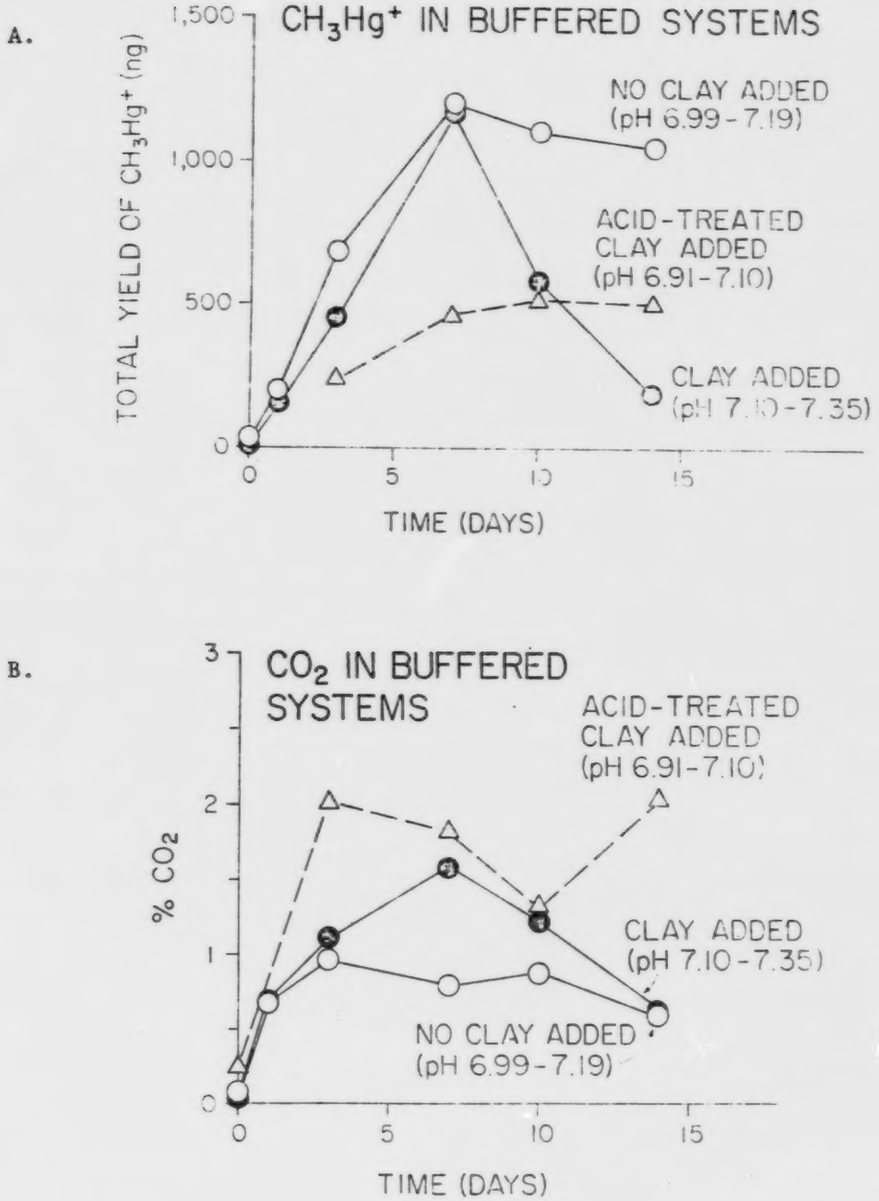

Fig. 21. Effects of untreated and acid-leached shoreline clay on production and destruction of CH_3Hg^+ and CO_2 by East Mynarski Lake sediment incubated under N_2 in the presence of $HgCl_2$, nutrient substrate (moss), and buffer ($CaCO_3$). Explanation of symbols: Same as in Fig. 20.

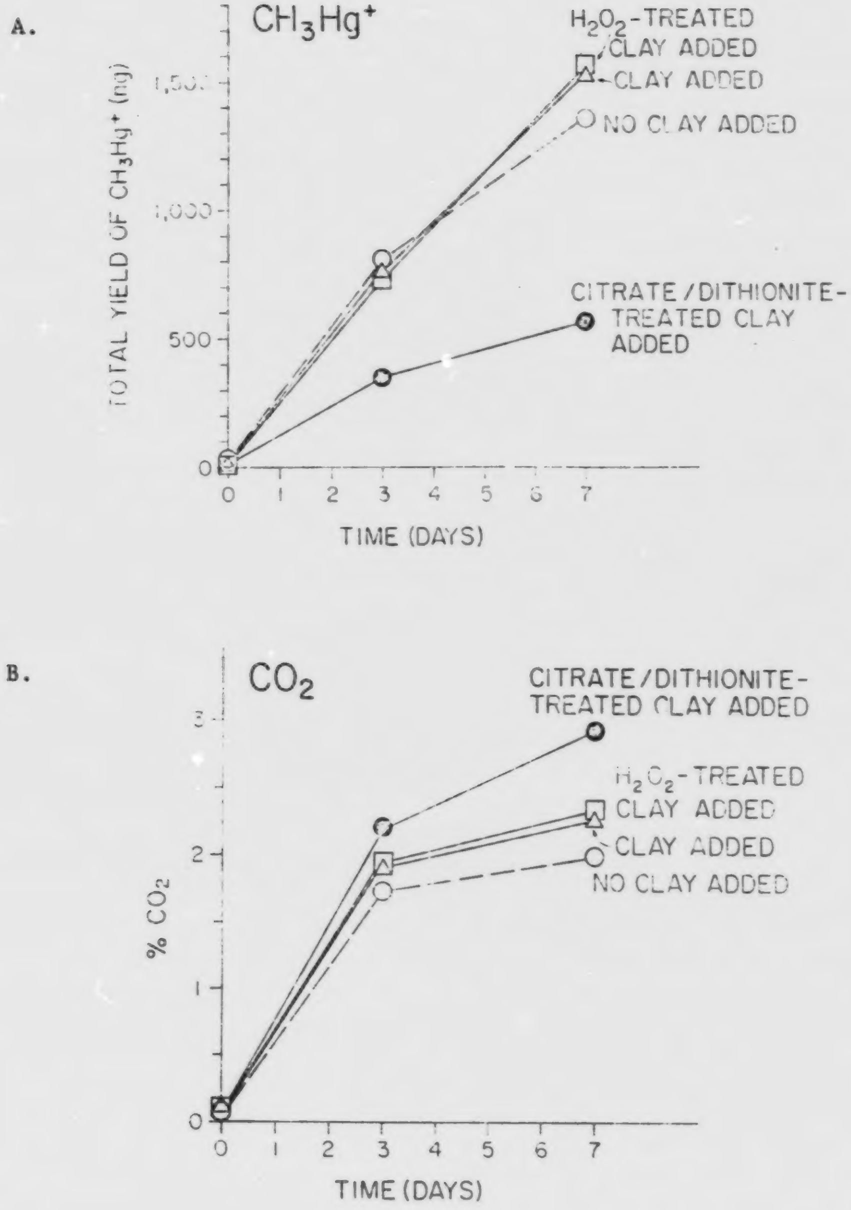

Fig. 22. Effects of shoreline clay, with and without H_2O_2 and citrate/dithionite pretreatments, on the production of CH_3Hg^+ and CO_2 by East Mynarski Lake sediment incubated under N_2 in the presence of $HgCl_2$, nutrient substrate (moss), and buffer ($CaCO_3$). Explanation of symbols: Control, ○ ; "untreated" clay added, △ ; H_2O_2-treated clay added, □ ; citrate/dithionite-treated clay added, ● .

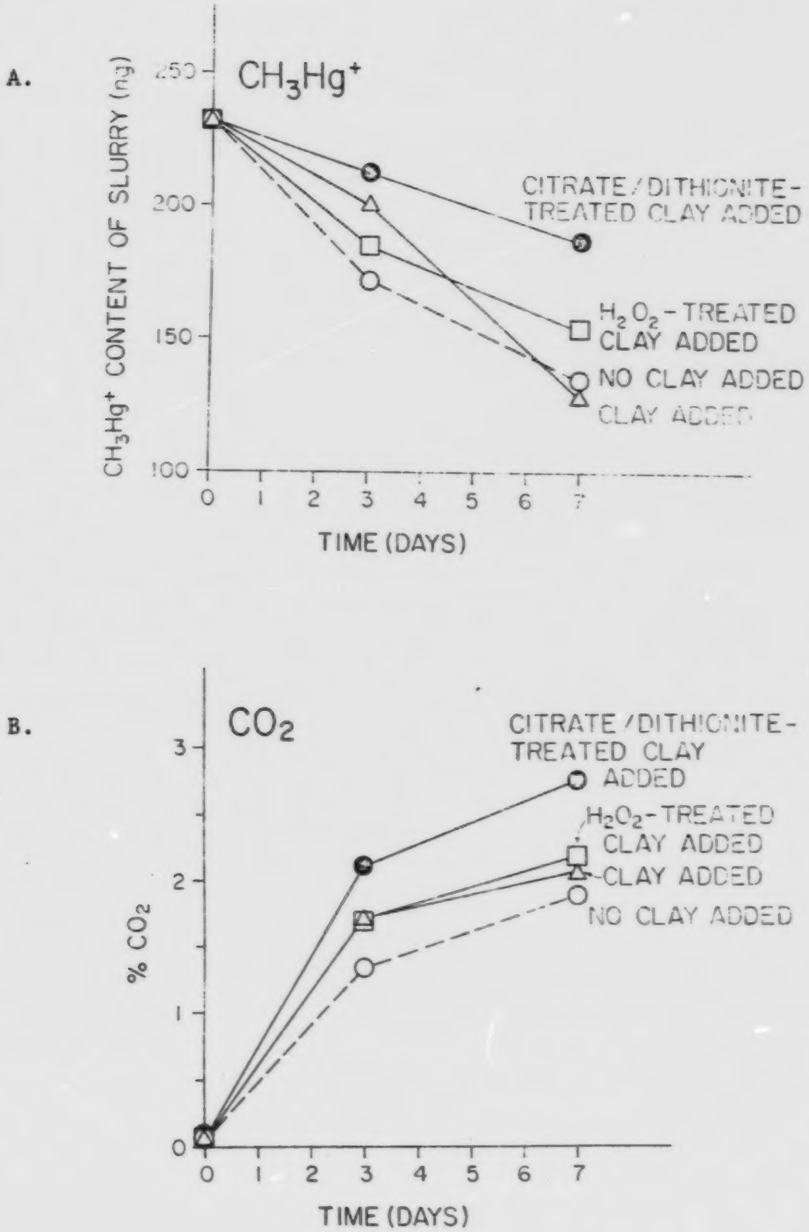

Fig. 23. Effects of shoreline clay, with and without H_2O_2 and citrate/dithionite pretreatments, on demethylation of CH_3Hg^{+2} and concomitant CO_2 production by East Mynarski Lake sediment incubated under N_2 in the presence of CH_3Hg^+ acetate, nutrient substrate (moss), and buffer ($CaCO_3$). Explanation of symbols: Same as in Fig. 22.

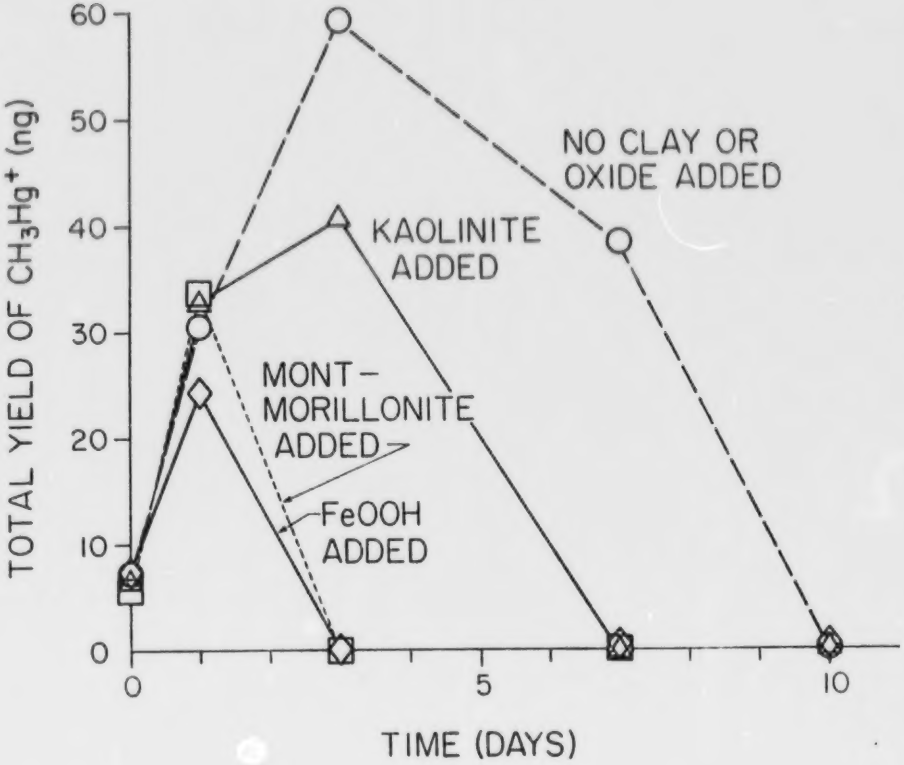

Fig. 24. Effects of kaolinite, montmorillonite, and FeOOH on production and destruction of CH_3Hg^+ by East Mynarski Lake sediment incubated under N_2 in the presence of $HgCl_2$, nutrient substrate (moss), and buffer ($CaCO_3$). Explanation of symbols: Control, ○ ; kaolinite added, △ ; montmorillonite added, □ ; FeOOH added, ◇ .

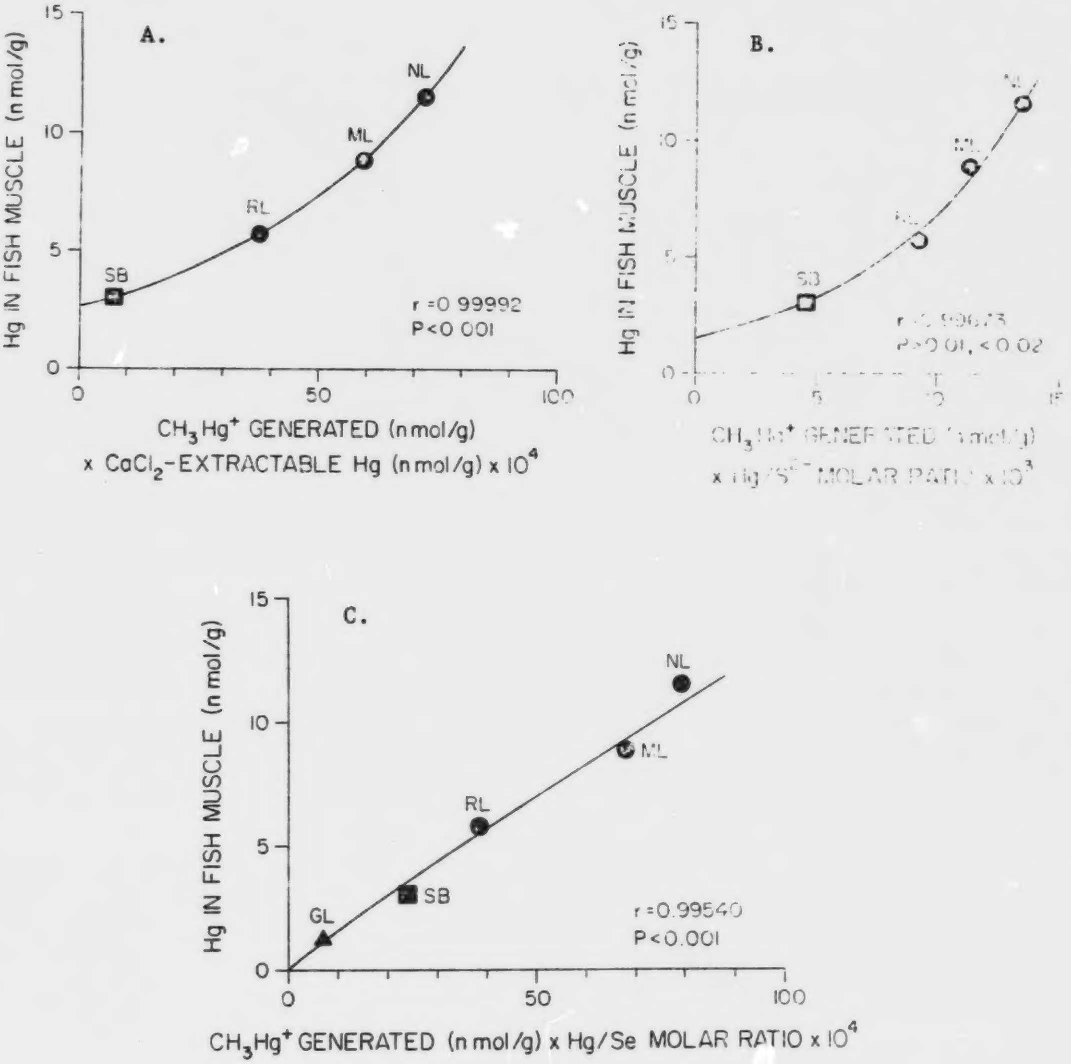

Fig. 25. Relationship between the Hg content of walleye muscle (mean for 1980 and 1981) and the "methylating capability" of bottom sediment collected in August, 1983 (CH$_3$Hg$^+$ data in Table 1) multiplied by each of three indexes of the abundance or relative abundance of "available inorganic Hg" in the sediment (representing samples collected in August, 1981 or August, 1983). Explanation of symbols: Granville Lake, ▲ ; Southern Indian Lake, ■ ; and Notigi Reservoir lakes, ● . Abbreviations: Granville Lake, GL; South Bay, SB; Rat Lake, RL; West and Central Mynarski Lakes, ML; Notigi Lake, NL.

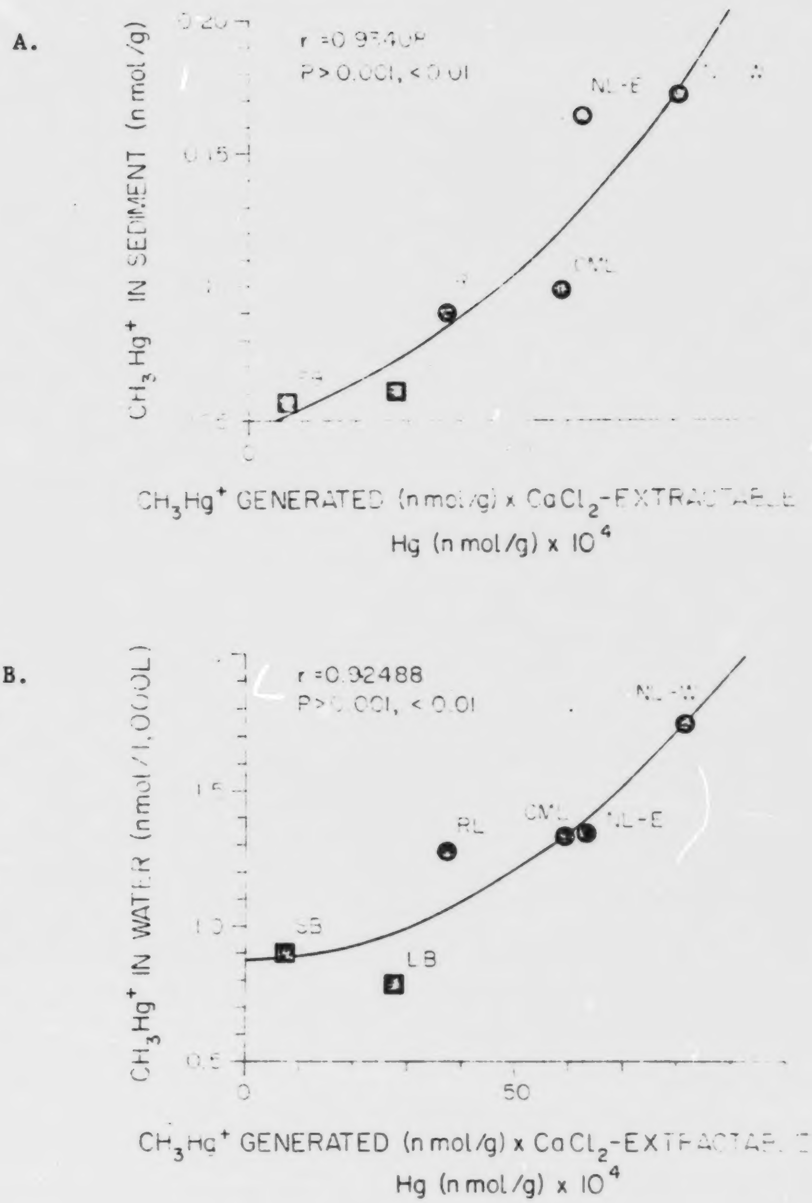

Fig. 26. Relationships between environmental CH_3Hg^+ concentrations (in water and sediment collected in June, 1981) and the "methylating capability" of sediment collected in August, 1983 (CH_3Hg^+ data in Table 1) multiplied by the concentration of "available inorganic Hg" in the sediments (representing samples collected in August, 1981). Explanation of symbols and abbreviations: Same as in Fig. 25.

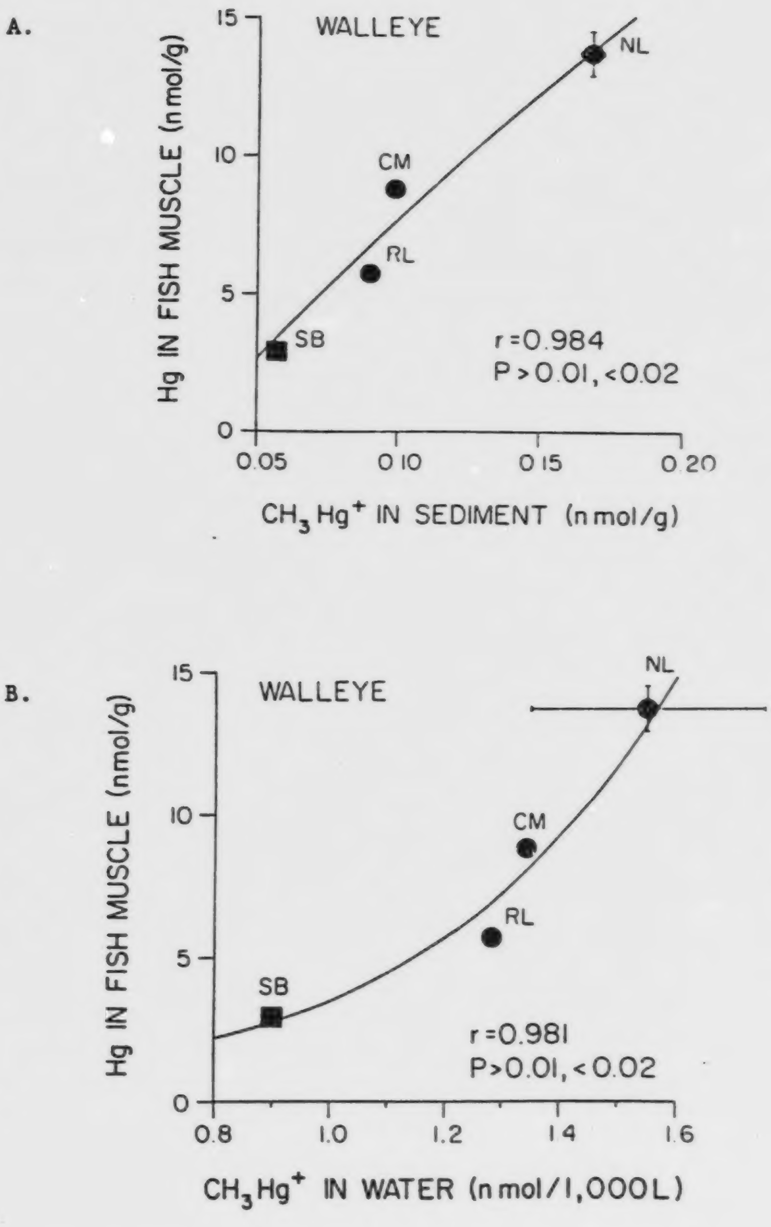

Fig. 27. Relationships between the Hg content of walleye muscle (mean for 1980) and the concentrations of CH_3Hg^+ in sediments and surface water (in June, 1981). Explanation of symbols and abbreviations: Same as in Fig. 25.

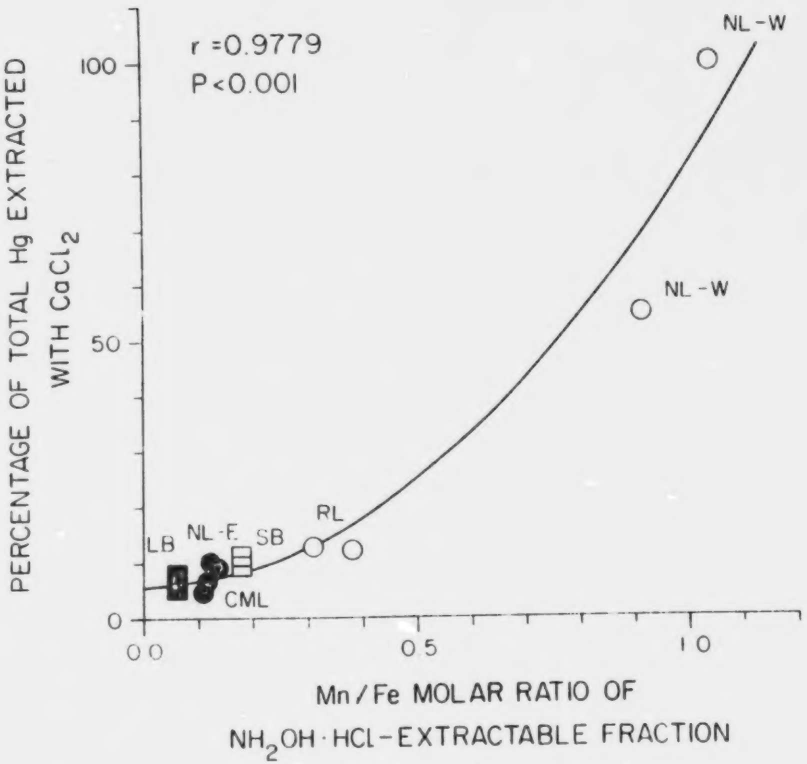

Fig. 28. Relationship between the relative availability of inorganic Hg (estimated as % extractable with $CaCl_2$ solution) and the Mn/Fe ratio of the $NH_2OH.HCl$-extractable oxide fraction in sediments collected in August, 1981. Explanation of symbols: Main-stem sites, white symbols; backwater sites, black symbols. Notigi Reservoir lakes ○, ● ; Southern Indian Lake □, ■ ; Abbreviations: Notigi Lake (west basin), NL-W; Notigi Lake (east basin), NL-E; Rat Lake, RL; Central Mynarski Lake, CML; South Bay, SB; Long Bay, LB.

Canada Manitoba

Canada-Manitoba Mercury Agreement

POSTSCRIPT PROJECT 2.3

The Technical Advisory Committee expressed the concern that the extrapolation of results of laboratory experiments to field conditions must always be made with caution. This caution had particular relevance to the results of Project 2.3 because of the "artificial conditions" involved in some experiments, eg., high temperatures and long storage time of sediments.

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Technical Appendices to the Summary Report Volume 3

EVOLUTION OF LIMNOLOGICAL CONDITIONS, MICROBIAL METHYLATION OF MERCURY AND MERCURY CONCENTRATIONS IN FISH IN RESERVOIRS OF NORTHERN MANITOBA

A summary report for Project 2.4 of the Canada-Manitoba Agreement on the Study and Monitoring of Mercury in the Churchill River Diversion

by

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SPECIAL NOTE

This report was prepared for the Canada-Manitoba Agreement on the Study and Monitoring of Mercury in the Churchill River Diversion. Any conclusions, recommendations, or opinions expressed herein are those of the authors and are not necessarily the same as those of the Steering Committee for this Agreement.

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1 Limnological data for the Southern Indian Lake and Notigi reservoirs

2 McCullough, G.K. Shoreline evolution in the Southern Indian Lake reservoir. Department of Fisheries and Oceans. Contract Report. 11 p. + 4 Tables + 4 Figures

3 Ramsey, D.J. and P.S. Ramlal. 1986. Measurements of mercury methylation balance in relation to concentrations of total mercury in northern Manitoba reservoirs and their use in predicting the duration of fish mercury problems in new reservoirs. Contract Report. Agassiz North Associates, Winnipeg. 33 p. + 7 Tables + 12 Figures

4 Ramsey, D.J. and P.S. Ramlal. 1986. Measurements of rates of production and degradation of methyl mercury and concentrations of total mercury in Southern Indian Lake, Cedar Lake and Granville Lake, Manitoba; Results of a survey conducted in July and August, 1985. Contract Report. Agassiz North Associates, Winnipeg. 24 p. + 12 Tables + 13 Figures + 1 Appendix

5 Bodaly, R.A. and N.E. Strange. 1986. Methods and results for mercury in large fish from Southern Indian Lake and Issett Lake, 1975-1985. Department of Fisheries and Oceans. 3 p. + 5 Tables + 10 Figures

EXECUTIVE SUMMARY

The objective of project 2.4 was to determine if there is evidence of change in mercury concentrations in fish from Southern Indian Lake and Issett Lake (upper Notigi Reservoir) and of improvement in the limnological conditions thought to be contributory to the mercury problem. These studies made use of a limnological data base extending back to 1972 and mercury in fish surveys extending as far back as 1975. In the course of project 2.4, new methods of measuring specific rates of mercury methylation and demethylation were applied to the study area and to older reservoirs in northern Manitoba to determine how the methylation-demethylation balance was changed in reservoirs and how these processes evolved over time.

The greatest changes in the limnological conditions in Southern Indian Lake and Notigi occurred with impoundment and diversion. In Southern Indian Lake, impoundment occurred in 1976 and was completed in one open water season. In Notigi Reservoir, impoundment began in early 1974 and was completed by the spring of 1976. Limited diversion flows began in 1976 but full diversion was not operational until 1978. Dramatic changes in shoreline erosion and turbidity occurred in Southern Indian Lake while in Notigi lower dissolved oxygen concentrations and higher nutrient concentrations occurred with impoundment. By 1984 erosion rates at unstable sites in Southern Indian Lake are essentially unchanged since 1977-78, but in the northern portion of the lake stable shorelines were more frequent. Highest turbidities were observed just after impoundment, but there is little evidence of improvement since 1978-79. At Notigi Reservoir, very high nutrient concentrations and low oxygen concentrations were observed in 1974-75. Both these parameters improved rapidly with diversion in the mainstem portions of the reservoir because of flushing with well-oxygenated Churchill River water which is low in dissolved nutrients. There has been little change in Notigi Reservoir conditions since 1978-79, and the reservoir is still further from oxygen saturation than Southern Indian Lake because of higher rates of microbial decomposition. In general, both reservoirs cannot be said to be recovering rapidly from the conditions which impoundment created. Given the lack of trend over a six year period, 1978-84, there is no reason to predict that conditions will improve in the near future.

Detailed survey data on mercury concentrations in fish are available for walleye and northern pike for the post-impoundment period (1978-85), while survey data on lake whitefish are also available prior to impoundment (1975-85). Whitefish in all regions of Southern Indian Lake and Issett Lake showed a maximum in mercury concentrations between 1977 and 1980 and have subsequently had lower values. In some cases, the mercury concentrations are still above pre-impoundment values. In contrast, there is no clear trend common to all study regions for the piscivorous species. Maximum mercury concentrations for pike in Issett Lake occurred in 1984, eight years after diversion but most regions have shown little trend over time. Walleye from Issett Lake had experienced a significant decline in mercury concentrations since 1978, but mercury concentrations in walleye from other regions have shown no obvious trend over time. Fish mercury concentrations and trends appear to depend on the diet and behavior of the species. Benthivores and planktivores have lower mercury concentrations than piscivores, and among piscivores, offshore, bottom dwelling species, e.g. burbot, have lower mercury concentrations than inshore, shallow-dwelling species, e.g. northern pike. This pattern is consistent with the flooded areas being the primary and continuing source of methyl mercury to the system. The temporal pattern in whitefish indicated that the deeper offshore sediments may also have contributed to net methylation of mercury in the immediate post-impoundment period. Offshore conditions have apparently improved since 1980 at the latest. The temporal pattern in pike has shown no decrease in mercury concentrations over time suggesting that net methyl mercury production has been proceeding at a relatively constant or accelerating rate, higher than the natural situation since 1978. It is not possible from the available data for mercury in fish or general understanding of the mercury problem in reservoirs to predict when mercury concentrations in fish in these reservoirs will decrease to the pre-impoundment levels. Improvement will not occur until net mercury methylation rates return to pre-impoundment levels.

Specific rates of mercury methylation in the flooded terrain of reservoirs are consistently higher than in the littoral zone of control lakes or in offshore sediments deposited by lakes before impoundment. Specific rates of mercury demethylation in flooded terrain are not higher than in control lakes. Specific rates of mercury methylation observed over four years, 1982-85, in one flooded area of Southern Indian Lake, have not shown any significant tendency to decline. Specific rates of mercury methylation were as high in the flooded, boreal forest terrain of 40 year old Reindeer Lake as in the recent reservoirs. These disquieting facts suggest that very long periods of time may be required for extensively flooded boreal forest reservoirs to exhibit mercury concentrations in fish similar to most natural lakes. The inferred slow recovery of specific rates of mercury methylation in flooded terrain is consistent with the lack of trend in the fish data over seven years in Southern Indian Lake and Notigi and the lack of a significant improvement in limnological conditions over the same time period. The only reservoir in Manitoba, Cedar Lake, known to have recovered from high mercury concentrations in fish had low specific rates of mercury methylation in 1985.

However, this reservoir which recovered over a period of 12-15 years, flooded largely marsh vegetation rather than boreal forest; and, therefore, cannot serve as a model for recovery for the reservoirs in the Churchill River Diversion area.

Further study is required to better elucidate the time course of elevated mercury concentrations in boreal forest reservoirs. In this regard, it is recommended that the monitoring program on Southern Indian Lake, which includes the longest available record on mercury concentrations in fish, specific rates of mercury methylation in flooded terrain and limnological conditions be continued until mercury concentrations in fish decline to pre-impoundment levels.

No environmental mitigation can be recommended at this time. Resuspension of sediments was not found to be effective in reducing mercury concentrations in fish in the Churchill River Diversion in the short to intermediate term. Increased understanding of the determinants of the balance between microbial methylation and demethylation may offer the best long term solution to the problem. If the species ecology of the microbes which methylate and demethylate were better understood, it might be possible to enhance demethylation activity or decrease methylating activity through the application of appropriate substrates. However, this would require basic research beyond the scope of the present study.

INTRODUCTION

The mercury concentrations in fish from Southern Indian Lake began to rise soon after the lake's impoundment in 1976 to effect Churchill River Diversion (Bodaly and Hecky 1979; Bodaly et al. 1984a). Notigi Reservoir was constructed between 1974 and 1976 to control discharge down the Rat and Burntwood Rivers diversion route. By 1978 efforts to open commercial fishing in Notigi Reservoir were curtailed because of unacceptably high mercury concentrations in the predatory species (Bodaly et al. 1984a). By 1979, fish from lakes downstream of Notigi Reservoir on the Burntwood River, where water levels were increased by the Churchill River diversion, were also showing increased mercury concentrations. Recognition of the real and potential problems caused by increased mercury concentrations in fish from lakes affected by the Churchill River Diversion led to a claim under the Northern Flood Agreement and to the signing of the Canada-Manitoba Agreement on the Study and Monitoring of Mercury in the Churchill River Diversion. This report summarizes the activities, results and conclusions of Project 2.4, Limnological Surveys, of the latter agreement.

The objective of Project 2.4 was "to relate the results of the other projects under this Agreement [to] the present physical condition of Southern Indian Lake and Notigi Reservoir" by studying "those aspects pertinent to the understanding of the cause and effects of mercury in the environment". The rationale for the project was predicated upon the availability of limnological records for Southern Indian Lake and Notigi Reservoirs collected by the Department of Fisheries and Oceans from 1972 to 1979, the period of impoundment and reservoir formation. As initially formulated, the project was to monitor the same limnological factors over the period of the agreement. It was assumed that recovery to pre-hydroelectic development limnological conditions would be the best guide to the return of fish mercury concentrations to pre-impoundment levels. However, it was recognized in the original project description that studies sponsored by the Agreement might discover new factors and techniques more appropriate to the objective of the project. This in fact happened, and the breadth and frequency of limnological observations was reduced over the term of the agreement, and greater emphasis was placed on field measurements of mercury methylation and demethylation by microbes. These measurements were made in Southern Indian Lake, Notigi Reservoir, older reservoirs in northern Manitoba and natural lakes in order to elucidate the time course of elevated fish mercury concentrations in these reservoirs.

In the original study agreement, the study and reporting of the fish mercury data from Southern Indian Lake were to be funded and reported under project 3.3, Fish Surveys and Fish Eating Guide. However, it was eventually recognized that the record on fish mercury concentrations in Southern Indian Lake was unique, i.e., extending from before impoundment, and was especially relevant to determining the time course of the mercury problem and predicting the eventual recovery of the system. Therefore, in the latter part of the term of the Agreement, the studies on fish mercury concentrations in Southern Indian Lake were funded through project 2.4, and all of the results of survey fishing for Southern Indian Lake and Issett Lake are reported here.

This report summarizes data and reports contained in five appendices: 1) all the relevant limnological data for these systems (Appendix 1), 2) a report on shoreline evolution in Southern Indian Lake (Appendix 2), 3) a report on the microbial mercury methylation-demethylation balance in reservoirs of different ages including Southern Indian Lake and Notigi Reservoirs (Appendix 3), 4) a similar report examining the methylation-demethylation balance in Cedar Lake, Manitoba (Appendix 4), and 5) all the survey data on mercury in fish in Southern Indian Lake and Issett Lake (Appendix 5). Tables and figures in these appendices will be referred to by adding the letter A and the appropriate appendix number to the table or figure number in the report. For example, Table 1 of Appendix 3 will be referred to here as Table A3.1.

PROJECT RATIONALE

Other studies, published as part of this agreement (Bodaly et al. 1987a) and elsewhere (Bodaly et al. 1984a), have established that fish from the Southern Indian Lake and Notigi Reservoirs have elevated mercury concentrations relative to fish from pre-impoundment Southern Indian Lake or to other unflooded lakes in the vicinity. The data, now covering a longer time period, on mercury concentrations in fish to be presented in this report further corroborate these other studies. The hypothesis that the flooding of terrestrial materials can cause fish to have these higher mercury concentrations has been experimentally tested and accepted (Hecky et al. 1987). Despite the elevated mercury concentrations in fish muscle, mercury either as total mercury or methyl mercury in water is generally below detection levels (nominally $<5\ \mathrm{ng \cdot L^{-1}}$ and $<0.2\ \mathrm{ng \cdot L^{-1}}$ respectively). Consequently it has not been possible to follow or model the evolution of the mercury problem using analyses of mercury in water. However, the hypothesis of project 2.4 is that water quality conditions created by reservoir formation are related to the problem of elevated concentrations of mercury in fish, and the corollary would be that mercury concentrations in fish would decline to pre-impoundment conditions when or soon after water quality is once again similar to the pre-impoundment condition. Water quality is used here in a general sense including any aspect of water or sediment (physical, chemical or biological) which influences the concentration of chemical compounds in the water or organisms in the water. The first section of this report reviews the limnological evolution of the Southern Indian Lake and Notigi reservoirs emphasizing those factors which have demonstrated or inferred relevance to the problem of mercury in fish. The second section discusses the balance of mercury methylation-demethylation in northern Manitoba lakes and reservoirs. Finally, the temporal record of mercury concentrations in fish particularly in Southern Indian Lake where the longest record exists will be reviewed as well as data on mercury concentrations in fish from several older reservoirs in northern Manitoba.

RESERVOIR EVOLUTION

Physiography

Newbury et al. (1984) reviewed the formation and physical characteristics of the Southern Indian Lake and Notigi reservoirs (Table 1). The Southern Indian Lake reservoir was formed in 1976 when the lake level was raised 3 m above its long term mean level in order to divert up to 850 $m^3 \cdot s^{-1}$ through a gravity fed diversion channel from the Churchill River basin to the Nelson River basin. The diversion channel connects South Bay of Southern Indian Lake with the uppermost end of the Notigi Reservoir near the former Issett Lake. The Notigi reservoir was developed behind a control structure on the Rat River where it flowed out of the former Notigi Lake. This dam controls discharge out of Notigi Reservoir and indirectly flows through the diversion channel. Water levels in the Notigi forebay were raised 15 m over the long term mean level of Notigi Lake.

The extent of flooding and the filling sequences were quite different for these two reservoirs although the flooded terrains were similar and typical of the nortnern boreal forest (Newbury et al. 1984). Southern Indian Lake was a large, shallow lake before flooding; and impoundment only increased the surface area from 1977 km^2 to 2391 km^2 and its mean depth from 8.5 to 9.8 m. Notigi Reservoir amalgamated seven, small shallow lakes with a combined area of 153 km^2 and average mean depth of 3.2 m to form a new water body of 584 km^2 and 7.8 m mean. Drawdown has not been a significant feature of either reservoir as neither is operated as a storage reservoir. Although the flooded areas are similar in both reservoirs, the ratios of reservoir volume to area of flooded land are quite different, 56.5 m and 9.5 m respectively. This ratio illustrates the potential dilution of any short term release of substances from flooded terrain in these two systems. Because the rate of mercury methylation is strongly peaked with highest values in late July and August and very low values the remainder of the year (Appendices 3 and 4), the higher dilution possible in Southern Indian Lake may explain the lower mercury concentrations in large predatory fish in Southern Indian Lake relative to Notigi noted by Bodaly et al. (1984a).

Southern Indian Lake was filled during July-September 1976 while the combined discharge at the natural outlet, Missi Falls, and the diversion channel was reduced. The Notigi Reservoir was filled over two years, starting in early 1974, by cutting off the flow of the Rat River to collect local runoff. This allowed the two reservoirs to be at nearly the same level when diversion flow was initiated at the end of June 1976. Diversion flows were then held between 200 and 400 m^3 during the remainder of 1976 until mid-1977 when diversion flows were increased to 800 $m^3 \cdot s^{-1}$. Since then, both systems have been very well flushed with Southern Indian Lake having a mean water exchange time of 260 days based on its whole volume while the exchange time of Notigi is 62 days. Such flushing times are somewhat deceiving as both reservoirs have complex outlines with substantial areas where flushing is slow (greater than one year exchange time) and determined by local runoff rather than the flow of the Churchill River (Newbury et al. 1984).

Southern Indian Lake and Notigi Reservoirs also differed in the extent and intensity of shoreline erosion. Southern Indian Lake has long fetches and a large proportion of permafrost-affected, glacio-lacustrine clay shorelines. Consequently, when impoundment raised water levels, shoreline erosion was extensive and rapid in many parts of the lake (Newbury and McCullough 1984) and high suspended sediment concentrations resulted (Hecky and McCullough 1984). In contrast most of Notigi Reservoir had short fetches, little shoreline erosion, and low suspended sediment concentrations (Bodaly et al. 1984b).

The differences in fetch also affected different regimes of thermal stratification in the two reservoirs. Before and after impoundment, Southern Indian Lake was well mixed (Hecky 1984) although particularly calm and warm summers in 1984 and 1985 allowed some thermal stratification in the deepest parts of the reservoir. In contrast, wind mixing is not as effective in Notigi Reservoir because of the limited fetches. Thermal stratification is common, especially in the southern portions of the reservoir where the greatest depths occur. Thermal stratification was more common during the filling period when there was no outflow. With the onset of diversion flows in 1976 and especially with full diversion flow starting in 1977, thermal stratification was largely eliminated in the mainstem portions of the reservoir (Bodaly et al. 1984b).

Limnology

The limnological features chosen for discussion, water temperature, conductivity, suspended solids, oxygen and phosphorus, are those shown to have relevance to the mercury concentrations in fish muscle by studies under Project 1.3 (Bodaly et al. 1987a) and/or Project 2.1 (Hecky et al. 1987) of the Agreement. Bodaly et al. compared mercury concentrations in small fish, in particular yellow perch and spottail shiners, from 13 stations scattered over the Churchill River Diversion area with values for chemical and physical parameters collected at the same stations during the same season (summer 1981). Small fish were used on the assumption that their limited mobility relative to the larger species would mean that the limnological observations made at a station were relevant to fish caught at that station. Also, their small size would allow their mercury concentrations to adjust more rapidly to local conditions compared to larger fish with much larger body burdens. Hecky et al. (1987) recorded the same suite of limnological observations in the limnocorral experiments which established the effect of various treatments on the uptake of mercury by yellow perch. The limnocorrals offer certainty that the fish in the limnocorrals were exposed to the limnological conditions monitored.

Water temperature

Bodaly et al. (1987a) found a negative correlation between mercury in fish and water temperature of the mixed layer. The possible effect of water temperature could not be specifically investigated in the limnocorrals. Although seasonal mean water temperatures in the limnocorrals did vary by 2°C over the four years of experiments, there was no obvious relationship between temperature and mercury concentration in fish in a stimulatory treatment of moss-peat material applied in every year. The highest mercury uptake did occur in a warm year and the lowest in a cool year which is counter to the correlation observed with the field data. Hecky (1984) observed that maximum water mass temperatures were depressed by as much as 2°C by the impoundment of Southern Indian Lake and the diversion of the Churchill River, and the effect on maximum surface temperatures was even greater (Patalas and Salki, 1984). The effect was greatest in the northern regions of the lake. Surface temperatures are highly dependent on whether or not the lake stratifies and quite high mixed layer temperatures were recorded in the northern portions of Southern Indian Lake in 1984 and 1985 when these areas stratified (Fig. A1.1 and A1.2).

Although the lower mixed layer temperatures in Southern Indian Lake early in the post-impoundment and diversion period and the coincident rise in fish mercury does support the inverse relationship found by Bodaly et al. (1987a), it cannot account for the substantial differences between mercury concentrations in larger fish species in Notigi and Southern Indian Lake. For example, the observed peak mercury concentrations in walleye in the Notigi Lake portion of Notigi Reservoir were 4-5 times as high as the highest values in Southern Indian Lake. The summer water temperatures in Notigi Reservoir were slightly higher than in Southern Indian Lake before and after diversion (cf. Fig. A1.3 with A1.1 and A1.2), and they show much the same inter-annual trends as Southern Indian Lake. Given the lack of any consistent relationship between water temperature fish mercury, it appears that the correlation found by Bodaly et al. (1987a) was fortuitous.

Conductivity

Bodaly et al. (1987b) found a significant positive correlation between the conductivity of water and mercury in fish. Conductivity is generally well correlated with total dissolved salts. Although the effect of conductivity on mercury mobilization was not specifically tested in the limnocorrals, the limnocorral treatments did effect significant changes in water conductivity. For example, moss-peat additions to limnocorrals lowered conductivity while suspended sediment and prairie sod additions raised conductivity. All three of these treatments led to increased mercury uptake in yellow perch. Conductivity per se was not a major factor in determining mercury concentrations in fish in limnocorrals and at most, can only be a modifying factor, not a causative factor, in elevated fish mercury concentrations.

Conductivity is a good "tag" for Churchill River water in the Churchill River diversion because the river tends to be low in conductivity relative to other local rivers and streams, for example the Rat River before impoundment (Bodaly et al. 1984b). Because the Churchill River is the major water source for the reservoirs, conductivity becomes an indirect estimator of flushing time. Regions with conductivity close to the conductivity of the Churchill River are assumed to be well flushed while regions with higher conductivity are considered poorly flushed (Table A.1.1). Consequently, if rapid flushing, resulting in high dilution, is important in preventing high methyl mercury concentrations in water and fish then positive correlations among water exchange time, conductivity and mercury in fish would be reasonable. Dramatic changes in conductivity occurred in the mainstem portions of Notigi Reservoir when diversion began. Unfortunately, there are no data on mercury in fish from this system until one year after full diversion was in effect. There is thus no direct evidence of the positive effect of flushing. The patterns of conductivity in the present reservoir have been stable since 1978 when the record on mercury in fish from the Notigi Reservoir begins.

Suspended sediments

Hecky et al. (1986) established that eroded bank clay stimulated the uptake of mercury by fish when added to a limnocorral with a natural sediment bottom. In no experiment did suspended sediments depress the uptake of mercury by fish. In Southern Indian Lake shoreline erosion was extensive, and suspended sediment concentrations rose after impoundment. The mercury problem may be aggravated as long as shoreline erosion continues to add suspended sediments to the lake in significant amounts. Suspended sediments effect visibility of the Secchi disk, and thus Secchi disk depths are good estimators of suspended sediment concentration at Southern Indian Lake (Hecky and McCullough 1984). Long term records of summer Secchi disk values indicate some improvement since the extremely low, early post-impoundment values of 1977-78, but none of the Southern Indian Lake regions with significant input of suspended sediments have returned to their pre-impoundment clarity (Fig. A1.4 and A1.5).

Newbury and McCullough (1984) forecast a very long period for restabilization of shorelines, on the order of 35 years to restore 90% of the shorelines to their pre-impoundment condition. Hecky and McCullough (1984) estimated a further 10 years for erosion of unstable nearshore deposits and return of lake turbidity (and Secchi disk values) to the pre-impoundment values. Analysis of pre-impoundment and post-impoundment aerial imagery, done under the sponsorship of this agreement has verified the erosion model of Newbury and McCullough (1984) on a new, independent set of shorelines. Newbury and McCullough (1984) estimated a mean erosion index (volume eroded divided by incident wave energy) of 0.00035 for Southern Indian Lake clay shorelines while McCullough (Appendix 2) estimated 0.00038 for the new clay shorelines studied in 1983. The constancy of this erosion index over time indicates that erosion will decline and stop only when bedrock is reached as Newbury and McCullough (1984) had assumed in their estimate of the recovery period. McCullough (Appendix 2) also found that on average 50% of the eroded material from newly studied clay shorelines was retained within 100 m of shore confirming the observation of Hecky and McCullough (1984) that a large proportion of the eroded material was deposited nearshore.

Recent erosion studies, therefore, largely confirm that decades will be required to return the Southern Indian Lake shorelines and turbidities to the pre-impoundment state. However, there will be regional differences in the recovery time required. McCullough (Appendix 2) found that a high proportion of eroded shorelines in the large northern basin of Southern Indian Lake had reached bedrock by 1984. In contrast, in the southern regions of the lake where wave-energies are lower and clay deposits more extensive, many shorelines are still eroding. Because eroded suspended sediments can cause increased mercury concentrations in fish, the elevated fish mercury concentrations in at least the southern regions of Southern Indian Lake could persist for a long time.

Suspended sediment concentrations in the Notigi Reservoir are high only in the northern mainstem portion where the turbid outflow from South Bay of Southern Indian Lake enters through the diversion channel. These concentrations are not higher than those occurring in Southern Indian Lake, and sedimentation in the upper reservoir rapidly clarifies the diversion channel inflow. In the Notigi forebay, post-impoundment Secchi disk depths are actually greater than pre-impoundment values as the former Rat River was quite turbid (Fig. A1.6). Other factors must account for the much higher mercury concentrations in fish in the Notigi Reservoir.

Oxygen

The limnocorral experiments indicated that terrestrial organic materials such as moss-peat, spruce boughs and prairie sod stimulated methyl mercury production and accumulation in fish. The two treatments relevant to the boreal forest, moss-peat which is the predominant ground cover and spruce which is the dominant tree, stimulated bacterial growth and oxygen consumption. Limnocorrals receiving these treatments had oxygen concentrations well below saturation with respect to the atmosphere. Analysis of these limnocorral experiments indicated that only very small amounts of the total mercury added with these materials entered the mercury cycle in one summer season. The decay of the added organic material may be very slow if the rate of liberation of mercury (≈1% per year, Hecky et al. 1987) is proportional to decomposition of the organic material which binds it. Because microbial decomposition consumes oxygen when it is available, oxygen conditions in new reservoirs may be indicative of the accelerated net methylation of mercury by bacteria in reservoirs.

Impoundment had no detectable effect on oxygen saturation in Southern Indian Lake as the lake was always near saturation (Figs. A.1.7 and A.1.8). Presumably the large volume of the lake and its rapid mixing by winds acting over large fetches kept it well oxygenated despite the inundation of large tracts of boreal forest. In contrast, in the east and west basins of Notigi Lake where water levels rose over 15 m, oxygen values in surface waters were reduced well below saturation during and after the period of reservoir filling (Figs. A1.9 and A1.10). Anoxia occurred in the bottom waters of Notigi during the filling period and persisted in the more poorly flushed basins even after diversion flow began to improve conditions in the mainstem of the reservoir. Improvements in deep water oxygen conditions were already apparent by 1978 in the east basin of the Notigi forebay (Bodaly et al. 1984b). Notigi Reservoir, although improved in degree of undersaturation of oxygen, still exhibits a net consumption of oxygen as water from Southern Indian Lake passes through it (Table A.1.1) and all areas of Notigi reservoir exhibit substantially lower oxygen values (Table A.1.1) than the water entering from South Bay (Region 6 of Southern Indian Lake).

The higher rates of bacterial oxygen consumption induced by the more extensive flooding and the more quiescent conditions in the Notigi Reservoir have resulted in lower oxygen concentrations there. Because pre-impoundment conditions were known and the oxygen concentrations in the major source water, the diversion flow from South Bay were also known, it is possible to confirm that bacterial activity was higher per unit of surface area in Notigi than in Southern Indian Lake. In limnocorral experiments, lowered oxygen concentrations were always associated with higher mercury uptake by yellow perch. Therefore higher rates of net methylation of mercury and accumulation of mercury in fish would be expected in Notigi than in Southern Indian Lake as was observed (Bodaly et al. 1984a).

Oxygen may be the most easily measured parameter with which to monitor the rate of recovery of Notigi reservoir. However, it may not be sufficiently sensitive to be an adequate indicator for net microbial methylation. It only will be useful in Notigi as long as the rates of consumption are much higher than the rate of diffusion of oxygen from the atmosphere. When these rates are similar or the rate of atmospheric exchange higher, then the oxygen values will not reveal whether oxygen consumption rates are elevated over the pre-impoundment condition because oxygen concentrations will always be near saturation. This has been the case in Southern Indian Lake before and after impoundment. Furthermore, if only a subset of the total microbial community is responsible for methylation of mercury, then a variable such as oxygen which reflects the activity of the whole aerobic community may not specifically indicate net microbial methylation. For example, anaerobic bacteria could methylate mercury without directly affecting oxygen concentrations in overlying water. The need for a specific measure of microbial methylation led to the development of techniques for estimating the microbial methylation-demethylation balance in new reservoirs to be discussed below.

Phosphorus

Messier et al. (1985) have suggested that phosphorus might be used as an index of decomposition in new reservoirs. Phosphorus is more convenient to interpret in terms of a mass balance than oxygen in that it avoids the uncertainties associated with the atmospheric equilibration of oxygen. Also it has the advantage that existing models (e.g. Grimard and Jones 1982) of the evolution of phosphorus concentrations in new reservoirs could be used to predict the time course of elevated Hg in fish. The time course model is based on the difference between observed phosphorus concentration in a new reservoir and the phosphorus concentration expected at equilibrium.

Along the Churchill diversion, the phosphorus concentration in Churchill River inflow to Southern Indian Lake can be used as an upper bound on the equilibrium phosphorus concentration because most lakes and reservoirs retain some phosphorus. Consequently, after the stabilization period when the flooded terrain no longer releases phosphorus, concentrations will be dependent on the Churchill River input. The observed difference between spring phosphorus concentration at the diversion channel and the Churchill River inflow is indicative of the contribution of P from erosion and decomposition during the winter within regions 1, 2, and 6 of Southern Indian Lake. Similarly, the difference in phosphorus concentrations between Notigi outflows and diversion inflows is indicative of the phosphorus release from the Notigi Reservoir in winter. This simple analysis neglects local inflows which are less than 5% of Churchill River flow and on average have similar phosphorus concentrations to the Churchill River (Cleugh 1974).

Phosphorus concentrations rose sharply throughout the Southern Indian Lake and Notigi Reservoirs as their water levels rose (Figs. A1.11, A1.12, A1.13 and A1.14). At the start of diversion flows in 1976, total phosphorus concentrations were over two times higher in Notigi outflows than in diversion channel inflows from region 6 of Southern Indian Lake (Figs. A1.11 and A1.14). The concentrations in Notigi Reservoir fell rapidly with the onset of diversion flows. By winter of 1979 the concentrations in Notigi had stabilized relative to Churchill River inflows to Southern Indian Lake and diversion inflows to Notigi (Fig. A1.15). Phosphorus concentrations in Notigi outflows were 0.26 $\mu mol \cdot L^{-1}$ higher than Churchill River inflows in March 1984 as compared to 0.24 $\mu mol \cdot L^{-1}$ in 1979. Consequently, there is no evidence of evolution of phosphorus release within the reservoir over this time period. It appears that phosphorus release from flooded terrain in Notigi was positive and had stabilized by 1979. Any model dependent on the evolution of phosphorus concentrations to predict mercury concentrations in fish would predict a very strong peak and rapid decline at Notigi Reservoir. Such a rapid decline has not been observed. The rapid decline in phosphorus concentrations in Notigi Reservoir was due primarily to flushing. There has been negligible change in phosphorus release from 1979 to 1984 in Notigi Reservoir indicating that any further declines in phosphorus release and, by inference in the model of Messier et al. (1985), mercury concentrations in fish will likewise be very slow.

METHYLATION-DEMETHYLATION BALANCE

The mercury compound found in contaminated fish is usually described as methyl mercury, and is always reported in terms of the concentration of mercury it represents. Correctly, it is methyl mercuric ion $[CH_3Hg]^+$ attached to suitably negatively charged parts of receptive compounds in the tissue. When not so attached, e.g. as a halide, it is both water and lipid soluble, and readily passes through biological membranes. In any form it is one of the most toxic of mercury compounds to mammals. Within higher organisms it concentrates to a different extent in different tissues with most of the body burden being carried in muscle tissue where the methyl mercury becomes strongly bound to sulfhydryl groups in proteins. About 90% of mercury in fish muscle is generally present as this methyl compound (Westoo 1966; Huckabee et al. 1979; Hecky et al. 1987).

In the aquatic environment, methyl mercuric ion is only produced by microbes to any significant extent. Bacteria methylate only the mercuric ion, and some bacteria can demethylate methyl mercury producing methane and elemental mercury. Methyl mercury can be produced by most bacteria aerobically, e.g. as a misdirected synthesis of methionine (Landner 1971) and anaerobically via the acetate synthetase system or during synthesis of vitamin B-12 (Wood et al.1972). Because these pathways are shared by a large number of bacterial species, the capacity to methylate is wide spread in the aquatic environment. Jensen and Jernelov (1969) showed that microorganisms in lake sediments could methylate mercury and that the rate of production was correlated with microbial activity. Microbial demethylation of mercury requires the mercuric reductase enzyme (Furukawa and Tonomura 1973) which is specific for the demethylation reaction. The gene for this enzyme is found on plasmids, i.e. not on the bacterial chromosome. Pan-Hou et al. (1980)

demonstrated that the ability to demethylate could be transferred, via the plasmid, from one bacterial strain to another. In a transfer experiment, they also showed that that, although the parent strain (source of the plasmid) could not methylate mercury, the strain could methylate mercury when the plasmid was removed. Consequently, whether a bacterial strain methylates or demethylates is likely dependent on the presence or absence of the plasmid containing the gene for mercuric reductase. Because different strains methylate and demethylate mercury, both processes occur concurrently in aquatic ecosystems. What determines the balance of the two processes is poorly understood (Lexmond et al. 1976).

Ramsey and Ramlal (Appendix 3) have applied the methods of Furutani and Rudd (1980) and Ramlal et al. (1986) to examine the in situ balance between methylation and demethylation in northern Manitoba lakes and reservoirs. They found that reservoirs and especially the flooded terrain generally had much higher specific methylation rates than natural lake sediments. In contrast, specific rates of mercury demethylation were not higher in reservoirs than in natural lakes. Consequently, the ratio of specific methylation rate to specific demethylation rate (M/D) was consistently higher in reservoirs than in unflooded lakes. This agreed well with the observations of Bodaly et al. (1984a) and Bodaly et al. (1987b; and below) who found elevated mercury concentrations in the fish of northern Manitoba reservoirs. Bodaly et al. (1986) observed a good correlation between specific methylation rate at a site and the mercury concentrations in small fish collected from the same site. Ramsey and Ramlal (Appendix 3) found that methylation activity and the M/D ratio had a strongly peaked seasonal pattern with the highest rates occurring in late July and August in all the lakes and reservoirs examined. The most rapid uptake of mercury into fish in limnocorrals also occurred in August

(Hecky et al 1987). Ramsey and Ramlal (Appendix 3) found no relationship between sedimentary mercury concentration or water column mercury concentration as estimated by mercury concentration in zooplankton and specific methylation rates or the M/D ratio. Similarly, mercury uptake by fish in limnocorrals was not related to mercury concentration in bottom materials or even the amount of mercury available for methylation (Hecky et al. 1987). All these data taken together are consistent with the hypothesis that methyl mercury production in aquatic environments uncontaminated by gross industrial mercury pollution is under microbial control and that new reservoirs disturb the balance between methylation and demethylation by favoring the growth of bacteria which methylate mercury.

Ramsey and Ramlal (Appendices 3 and 4) have measured methylation and demethylation rates in lakes and in reservoirs up to 40 years old in northern Manitoba. Only two of those reservoirs did not show methylation rates which were high relative to the natural lakes. Sipiwesk Lake, behind the Kelsey generating station, was sampled during a low water stage which made it impossible to sample over flooded terrain. The low rates measured were done over fine-grained clay sediments. Therefore, the results in Sipiwesk are not comparable to the other reservoirs which were sampled over flooded terrain. Cedar Lake, formed behind the Grand Rapids generating station, had had elevated fish mercury concentrations in fish in the early 1970's, but the concentrations had declined over time until by the late seventies they were as low as natural lakes of the region (see discussion below). The low rates of methylation measured in Cedar Lake in 1985 are consistent with the present low mercury concentrations in fish in Cedar Lake.

Cedar Lake is probably not an appropriate model for the recovery of reservoirs formed by the Churchill River Diversion project because it flooded an extensive area of the sandy delta of the Saskatchewan River which had predominantly marsh vegetation. The other reservoirs studied were in boreal forest with peaty organic soils. The dynamics of organic degradation are likely quite different for these different vegetation and soil regimes. Methylation rates and the M/D ratios were as high in flooded areas of 14 year old Stephens Lake and 42 year old Reindeer Lake as in eight year old Southern Indian Lake. Enhanced methylation rates and elevated M/D ratios can persist in the flooded areas of boreal reservoirs for a very long time. Decades rather than years is the appropriate time scale for many northern reservoirs.

THE TEMPORAL RECORD OF MERCURY CONCENTRATIONS IN FISH

Evidence has accumulated rapidly in the past decade that elevated mercury concentrations in fish muscle often occur in newly constructed reservoirs. Three studies are now available which compare post-impoundment mercury concentrations in fish muscle to pre-impoundment concentrations. These results, from three different geographic areas, all revealed significant increases coincident with impoundment. Three lakes in northern Manitoba (Issett, Wuskwatim and Southern Indian) were studied by Bodaly et al. (1984a). In South Carolina, the Richard B. Russell Reservoir was studied by Abernathy et al. (1985) and in northern Quebec, Boucher et al. (1985) examined mercury concentrations in fish in the LG2 and Opinaca reservoirs. These studies all demonstrated higher mercury concentrations in fish after impoundment. Many other instances of high levels of mercury in fish after impoundment are known and it is often inferred that these high levels represent increases over (unknown) pre-impoundment levels. Examples include reservoirs in Labrador (Bruce and Spencer 1979), Finland (Lodenius et al.

1983; Verta et al. 1986), South Carolina (Abernathy and Cumbie 1977), Arizona (Potter et al. 1975), Idaho (Benson et al. 1976), Illinois (Cox et al. 1979), and Mississippi (Knight and Herring 1972).

Although studies invoking a rise in fish mercury concentrations in newly created reservoirs are numerous, empirical evidence for rates of return of mercury concentrations in fish in reservoirs to pre-impoundment levels is very scarce. Abernathy and Cumbie (1977) examined mercury concentrations in the muscle of largemouth bass from Lake Jocassee, South Carolina, one and two years after impoundment. The mean concentration in fish was significantly lower in the second year of sampling, and it was stated that the fish comprising the two samples were of similar size. However, these results are not convincing because sample sizes were quite small (n = 11 and 12) and a much longer time period than two years is desirable to define temporal trends. Cox et al. (1979) reported mercury concentrations in largemouth bass taken from Cedar Lake reservoir, Illinois, three and four years after filling. It was concluded by these authors that the data demonstrated a decline in mercury levels over the two years for which data were available. However, this conclusion must be considered to be tenuous because the sample size in the second year's was small (n = 9) and the range of mercury concentrations for the second year sample was within the range of values for the first year's sample.

There are two reservoirs in Manitoba for which fairly long, post-impoundment records of mercury concentrations in fish exist because these reservoirs were flooded before routine mercury testing began about 1970. These two reservoirs are Cedar and Sipiwesk lakes. Unfortunately, there are no pre-impoundment mercury data for either of these cases. Cedar Lake, located on the Saskatchewan River immediately before it flows into Lake

Winnipeg, was flooded in 1964 and 1965 by the Grand Rapids hydroelectric project. The area of Cedar Lake increased from 1352 km^2 to about 2400 km^2 (excluding Cross Bay) by flooding an extensive deltaic marsh. Records of the mercury concentrations in northern pike and walleye exist for the period 1971 to 1981, for both survey and commercial type samples (Department of Fisheries and Oceans, Winnipeg, MB, Southern Operations Directorate, unpubl. data). Mercury concentrations in fish from Cedar Lake in the early 1970's were elevated above those occurring in most natural lakes. These elevated mercury levels in fish probably were due to the flooding of Cedar Lake in 1964 rather than the downstream effects of a chlor-alkali plant located on the South Saskatchewan River at Saskatoon; Uthe et al. (1973) have shown that the effects of mercury contamination from the Saskatoon plant did not extend downstream as far as Cedar Lake. Concentrations in walleye showed a steady decline over the period 1971-1981, decreasing from about 0.55 $\mu g \cdot g^{-1}$ to about 0.20 $\mu g \cdot g^{-1}$. Concentrations in pike appeared to decline from about 0.85 $\mu g \cdot g^{-1}$ in 1970-71 to about 0.35 $\mu g \cdot g^{-1}$ by about 1976; they seem to have been relatively stable since then. Thus, in the Cedar Lake reservoir, mercury levels in predatory fish were declining 7 years after flooding and may have reached stable, lower levels about 16 years after impoundment. Even with the relatively long record for this lake, identification of these trends is difficult because of the presence of year-to-year variation in mercury concentrations of fish.

The case of Sipiwesk Lake provides little useful information on the recovery times of elevated mercury concentrations in fish in reservoirs. Sipiwesk Lake is located on the Nelson River between Split and Cross lakes and was flooded in 1965 by the Kelsey Generating Station Project. The level of the lake was increased by 1-2 meters. Again, no pre-impoundment data concerning mercury concentrations in fish exist. Mercury concentrations in fish are known for the period 1970-1984. Mean concentrations for northern pike and walleye have usually been in the range 0.6-1.0 $\mu g \cdot g^{-1}$ and show no discernible trend over the 15 years of record.

Many reservoirs which contain fish with elevated mercury concentrations are fished for commercial sale and/or for local consumption by people and dogs. Commercial shipments of fish are routinely monitored in Canada and sampled for mercury concentration. Sale of such shipments is regulated according to the permissible limits prevailing in the various countries of destination. No such routine sampling program exists for lakes and reservoirs which are fished for local consumption. It is therefore important to define mercury concentrations in fish consumed locally to determine if such consumption is a potential health risk.

The objectives of the study concerning mercury concentrations in the muscle of large fish species are therefore threefold:

1. to define mercury concentrations in the common domestically caught and consumed fish in the Southern Indian Lake and Issett Lake reservoirs;
2. to compare pre- and post-impoundment samples to determine whether increases in mercury concentrations have occurred;
3. to determine long term post-impoundment trends of mercury concentrations in fish.

This part of the report gives a detailed presentation and discussion only of survey type data for the areas noted above. The detailed results from monitoring of commercial shipments of fish are not included, but are mentioned briefly in the discussion below as a comparison to survey data. Methods and detailed results are to be found in Appendix 5.

Mercury Concentrations Reported

There were significant between year differences in the average size of whitefish, walleye and northern pike for almost all sample sites (Table A5.5). Therefore, two statistical methods, linear regression and analysis of covariance, were employed to adjust mean mercury concentrations for year-to-year differences in the average sizes of fish (Appendix 5). However, use of these procedures resulted in little change in mean mercury values in most samples (Tables A5.2-4) or in the identification of year-to-year trends in mean mercury concentrations (Figs. A5.2-10). This probably was due to the fact that the range in the average size of fish in the various samples was usually quite small (despite the fact that most differences were statistically significant) and because there were usually not significant correlations between mercury concentrations and fish size within individual samples (Bodaly et al. 1984a). Only mercury values obtained for the walleye samples from Issett Lake were changed noticeably by the two adjustment procedures. This was because the average sizes of fish in these samples were the most variable, ranging from 284 to 489 mm mean fork length, of any of the series of samples given in this report. It would appear that, if field sampling procedures are utilized which result in fish samples which do not vary markedly in average size from year-to-year, then statistical adjustment of the data will not result in dramatic changes in year-to-year trends.

Mercury Concentrations in Domestically Fished Species

There were noticeable between species differences in mean mercury concentrations. These differences were consistent, generally, between the various sites. Thus, in terms of simple mean muscle mercury concentrations, the species sampled could be ranked approximately as follows: northern pike >

walleye > burbot > cisco > whitefish = longnose sucker. In Southern Indian Lake, northern pike usually averaged near 0.65-0.70 $\mu g \cdot g^{-1}$, walleye near 0.60 $\mu g \cdot g^{-1}$, burbot near 0.25 $\mu g \cdot g^{-1}$, cisco near 0.20 $\mu g \cdot g^{-1}$ while lake whitefish and longnose suckers usually averaged near 0.10-0.15 $\mu g \cdot g^{-1}$ (Tables A5.1-4; Figs. A5.2, 5, and 8).

There also tended to be consistent differences between sites in the mean mercury concentrations of the various species sampled. Mercury concentrations in fish from Issett Lake were usually higher than concentrations in fish from any of the Southern Indian Lake sites. This was true for walleye (Table A5.2; Fig. A5.5), for lake whitefish (Table A5.1; Fig. A5.2), and for cisco (Table A5.4), while mercury concentrations in northern pike and longnose sucker were generally similar in Issett Lake and the Southern Indian Lake sites (Tables A5.3 and 4; Fig. A5.8). Within Southern Indian Lake, there were also noticeable differences in mercury levels in fish between sampling areas, although these differences were not as large as those often prevailing between Issett Lake and Southern Indian Lake. Thus, fish from Regions 6 and 5 tended to have the highest levels, while fish from Region 4, The Channel and Camp 9 have tended to have slightly lower levels (Tables A5.1-4; Figs. A5.2, A5.5 and A5.8). This generalization was not always true for a given species in a given year. Specifically, lake whitefish mercury concentrations were generally higher, during the post-impoundment period, in The Channel, Region 5 and Region 6, and were generally lower in Region 4 and Camp 9, walleye mercury concentrations were generally higher in Regions 6 and 5 and lower in The Channel, Camp 9 and Region 4, northern pike mercury concentrations were noticeably higher in Region 6 but lower in all other areas, and cisco mercury concentrations were generally higher in Region 6 and Camp 9, intermediate in Region 5 and lower in The Channel and Region 4 (Tables A5.2-4; Figs. A5.2, 5 and 8).

Comparisons of Pre- and Post-Impoundment Mercury Concentrations

The only species of fish from the Southern Indian Lake/Issett Lake area for which pre-impoundment survey samples are available is the lake whitefish. Significant increases in muscle mercury concentrations in lake whitefish took place from 1975, before lake impoundment, to 1978 and 1979, 2 to 3 years after impoundment (Table A5.1, Fig. A5.2). Increases took place in all areas sampled, i.e. The Channel, Camp 9, Regions 4, 5 and 6 of Southern Indian Lake and Issett Lake (Table A5.1). Differences in mean mercury concentrations in lake whitefish muscle from before impoundment to immediately after impoundment ranged from 0.08 $\mu g \cdot g^{-1}$ to 0.24 $\mu g \cdot g^{-1}$, as compared to mean mercury concentrations before impoundment of 0.05-0.07 $\mu g \cdot g^{-1}$. These increases represent increases of approximately 3-5x pre-impoundment concentrations (Table A5.1).

Post-Impoundment Trends in Fish Mercury Concentrations

Post-impoundment records of up to eight years in length are now available for the three major domestically fished species (whitefish, pike and walleye) (Tables A5.1, A5.2, and A5.3). The following discussion of temporal trends is based on mean mercury concentrations adjusted by analysis of covariance.

Mercury concentrations in whitefish at all sites sampled showed noticeable declines over the period 1978-79 to 1982 (Fig. A5.4). Statistical comparisons of mean mercury concentrations by analysis of covariance indicated significant year-to-year differences at all sites sampled (Table A5.5). Trends in mean concentrations are difficult to discern after about 1982, six years after impoundment. Since 1982, concentrations in whitefish were at or close to mercury concentrations found in 1975, before impoundment (Fig. A5.4). Concentrations in whitefish may have, therefore, returned to pre-impoundment levels by about six years after impoundment, although because only one year of sampling was conducted prior to impoundment, pre-impoundment levels were not very precisely defined. In Region 5, on the other hand, concentrations in whitefish did not appear to have returned to pre-impoundment levels by 1984.

In northern pike, year-to-year variation in muscle mercury concentrations was not significant at one site (SIL Camp 9), but was statistically significant at the $p<0.05$ level (one site) or the $p<0.01$ level (four sites) at other sites sampled (Table A5.5). Noticeable increases in pike mercury concentrations have taken place over the post-impoundment period at Issett Lake and in Southern Indian Lake Regions 6 and 4. A trend is difficult to identify for the SIL Region 5 site (Fig. A5.10).

Identification of any post-impoundment trends in walleye at the Southern Indian Lake sites is difficult. Analysis of covariance indicated that at one Southern Indian Lake site (The Channel), year-to-year variation in mean mercury concentrations was not significant, at one site, variation was significant only at the $p<0.05$ level, while at two sites, variation was significant at the $p<0.01$ level (Table A5.5). Therefore, year-to-year differences at the Southern Indian Lake site were slight compared to other species and obvious trends were not present at most sites (Fig. A5.7). There was a very slight indication of decreasing mercury levels at the Camp 9 site (Fig. A5.7). At Issett Lake, in contrast, there was a noticeable decline in mercury concentrations in walleye since impoundment (Fig. A5.7). Year-to-year variation in mercury levels in walleye from Issett Lake was significant at the $p<0.01$ level (Table A5.5).

The available data for other species (cisco, longnose sucker, and burbot) are not complete enough for the detection of temporal trends (Table A5.4), especially considering the year-to-year variability present in mean mercury concentrations in other species sampled.

Implications

Statistical analyses of mercury concentrations in fish muscle demonstrate that significant increases in the concentration of total mercury in lake whitefish have occurred after the impoundment of Southern Indian Lake (SIL) and the diversion of the Churchill River. Bodaly et al. (1984a) came to the same conclusion based on a more limited set of the data than is presented here. Included in Bodaly et al. (1984a) was an examination of the available commercial fish mercury monitoring data and it was concluded on the basis of these data that increases in fish mercury levels in walleye and northern pike had also taken place from before to after SIL impoundment. Bodaly et al. (1984a) noted that mercury concentrations in walleye were stable at 0.2-0.3 $\mu g \cdot g^{-1}$ over the period 1971-77 but were much higher after impoundment (1978-82) at 0.57-0.75 $\mu g \cdot g^{-1}$. Mercury concentrations in northern pike were in the range 0.25-0.35 $\mu g \cdot g^{-1}$ in 1971-73, appeared to be elevated immediately after lake impoundment in 1976-78 to 0.4-0.5 $\mu g \cdot g^{-1}$, and rose to 0.67-0.95 $\mu g \cdot g^{-1}$ in 1979-82. A more complete commercial sampling record for walleye and northern pike from Southern Indian Lake is now available. The following data is from the Department of Fisheries and Oceans, Southern Operations Directorate, Winnipeg, MB:

	Walleye		Northern pike	
	Mercury ($\mu g \cdot g^{-1}$)	Number of samples	Mercury ($\mu g \cdot g^{-1}$)	Number of samples
1971	0.19	6	0.26	4
1972	0.21	3	0.32	5
1973	0.28	3	0.30	3
1974		0		0
1975	0.30	2		0
1976	0.24	4	0.47	10
1977	0.26	2	0.43	2
1978	0.57	7	0.50	7
1979	0.75	6	0.88	9
1980	0.54	8	0.67	14
1981	0.62	34	0.91	39
1982	0.62	5	0.90	17
1983	0.49	3	0.81	26
1984	0.42	1	0.97	15
1985	0.32	1	0.81	1

The commercial records confirm the post-impoundment trends in mercury concentrations in walleye and northern pike which were seen in the survey data. Although commercial sampling and survey sampling are carried out by different methods (see McGregor (1987) for a detailed description of commercial sampling methods), the results usually show fairly close agreement (compare text table above with results in Tables A5.2 and 3; also see Bodaly et al. 1984a). Thus, the commercial results presented above show that mercury levels in northern pike appear to have increased steadily over the post-impoundment period, while walleye levels appear to be stable or slightly decreasing over the post-impoundment period (although commercial sample sizes for walleye are quite small in later years and conclusions based on these later data must remain tentative). It must also be recognized that commercial sampling does not control for fish size or specific area of the lake.

The present results demonstrate that elevated mercury concentrations in predatory fish in the Southern Indian Lake and Notigi Reservoirs did not decline rapidly as the reservoir aged, as would be predicted from other studies. Abernathy and Cumbie (1977) hypothesized that elevated mercury concentrations in fish in reservoirs was a relatively short-lived phenomenon, at least in South Carolina, lasting perhaps 3 to 5 years. Cox et al. (1979) hypothesized that mercury concentrations in fish in reservoirs should be elevated for only about five years after impoundment. These latter authors based their hypothesis on data which they interpreted to show a depletion of soil mercury in reservoir flooded zones in an Illinois reservoir three years after impoundment. However, Bodaly et al. (1987a) found little depletion of soil mercury six years after impoundment in northern Manitoba reservoirs. Abernathy et al. (1985) predicted a five- to eight-year period of elevated mercury levels in fish in a Piedmont reservoir but noted that this time course could be lengthened in the case of cold, oligotrophic impoundments. Present data on mercury concentrations in Southern Indian Lake fish do not allow predictions of the time required for a return to pre-impoundment concentrations, except to say that, with little indication of any post-impoundment decline in levels in predatory fish nine years after flooding, pre-impoundment concentrations probably cannot be reached for at least 15 or more years. These results are consistent with the observation that rates of mercury methylation at SIL did not decline over the period 1982-85, and that high rates of methylation occur in the flooded terrain of reservoirs up to 40 years in age (Ramsey and Ramlal, Appendix 3). Other factors which have been shown or inferred to affect or model mercury concentrations in fish, i.e. oxygen consumption, phosphorus release or suspended sediment concentrations are likewise showing little if any improvement over time in the affected reservoirs.

The relatively large amount of year-to-year variation in fish mercury concentrations demonstrates the need for a period of data collection which spans a number of years for the identification of temporal trends. This period should be at least four years in the case of mean mercury concentrations which show gradual monotonic temporal trends. Also, three years of records and relatively large samples would be required to define mean mercury concentrations for the fish of a lake basin if long term trends were not present. These data requirements have rarely been met in the case of other published studies.

The factors causing year-to-year variation in mercury concentrations in fish are not obvious. This variation may be related to annual variation in the amount of methyl mercury produced by bacterial methylation in flooded areas of the lake. Bodaly et al. (1987a) and Ramsey and Ramlal (Appendices 3 and 4), have shown that rates of mercury methylation over flooded terrain in the Southern Indian Lake reservoir tend to peak in August every year. This is usually the period of maximum water temperature. Also, Hecky et al. (1987) showed that August was the month of maximum uptake of methyl mercury by fish in limnocorrals. Therefore, variables which may have an effect on fish mercury concentrations in the Southern Indian Lake reservoir include water temperatures, as they could affect rates of mercury methylation (and demethylation) over flooded soils and terrestrial vegetation, water levels, as they affect the amount of soil and terrestrial vegetation (i.e. bacterial substrate) which is flooded from year-to-year, and shore erosion, as it affects the amount of terrestrial material input to the reservoir and the concentration of suspended sediment. These three factors should be acting in a manner which would result in changes in mercury concentrations in fish which are consistent between the various Southern Indian Lake sites. However, temporal variation is not consistent between sites, that is, there appeared to be no parallel pattern of mean mercury concentrations either within species between the different sites or at a given site between different species (Figs. A5.4, 7 and 10). A further possible factor which could be causing year-to-year variation in fish mercury levels is the month of the year (over the open water season) when sampling was conducted. However, an examination of the year-to-year pattern of mean mercury concentrations and the relative time of year that the various species of larger fish were captured at the various sites revealed no obvious correlations, despite the fact that the time of fish collection varied from June to early October.

Regional differences in fish mercury concentrations were largely consistent with the hypothesis of Bodaly et al. (1984a) that the relative amount of flooding is a dominant factor in determining post-impoundment mercury levels. Experimental testing of this hypothesis (Hecky et al. 1987) has established that the dominant organic materials of the boreal forest stimulate methyl mercury production and that stimulation is approximately proportional to the amount added. Fish from Issett Lake usually had the highest mercury levels and this site was the most extensively flooded of all those sampled (water level increased by 7.4 m). Region 5 was the most extensively flooded area of SIL (water level increased by 3 m resulting in a 45% increase in surface area) and fish from this area often had the highest mercury concentrations of the SIL sites. On the other hand, Region 6 was one of the least extensively flooded areas of SIL (16% increase in surface area) and fish from this area were usually higher than other, more extensively flooded sites on SIL. This latter observation could have been due to migration of some fish from Issett Lake into SIL Region 6, as these adjacent areas are now connected by the South Bay diversion channel. Because mercury concentrations were usually higher in fish from Issett Lake, migration of these fish into Region 6 would tend to elevate mean mercury concentrations in the latter area. Moreover, Region 6 also has one of the highest rates of shoreline erosion and suspended sediment concentrations (Hecky and McCullough 1984). Limnocorral experiments have shown that eroding bank clays in suspension also can increase mercury accumulation in fish (Hecky et al. 1987). Shoreline erosion also causes a continual input of terrestrial soil and vegetation to the aquatic system and these materials have been shown to increase mercury accumulation in fish.

Although little is known concerning the primary route(s) of uptake of methyl mercury by fish in lakes and reservoirs, the differences observed between the mercury concentrations and temporal response of mercury levels in the various species sampled in this study are consistent with known differences in habitat utilization and feeding patterns. Whitefish and longnose sucker are both deep-water benthivores and their mercury levels were usually quite similar. The three piscivorous species sampled, northern pike, walleye, and burbot, had the highest mercury concentrations. Piscivorous species almost always have higher mercury concentrations than non-piscivorous species, although the ratio of concentrations in piscivorous to non-piscivorous species can vary greatly (e.g. see Jones et al. 1986). Cisco, a planktivore, showed levels intermediate between the benthivores and predators. Piscivores were ranked, in terms of mercury levels, as northern pike > walleye > burbot, and this ranking corresponds to the relative degree of utilization of inshore habitat. This ranking is consistent with the observation that rates of mercury methylation (and the ratio of mercury methylation to mercury demethylation) are much higher over flooded (inshore) terrain in the Southern Indian Lake reservoir than over existing (offshore) lake sediment and that mercury levels in small fish which inhabit inshore areas are related to rates of mercury methylation (Bodaly et al. 1987a).

Differences among the three species for which post-impoundment trends in mercury concentrations can be described (northern pike, walleye, and whitefish) are also consistent with differences in the amount of utilization of inshore habitat. Thus, mercury concentration in northern pike appeared to be increasing nine years after SIL impoundment and this species utilizes inshore habitat to the greatest degree of the three species. Mercury concentration in whitefish have declined from peak levels reached 2-3 years after lake impoundment and this species utilizes primarily offshore, deepwater habitat. Walleye is intermediate with respect to both mercury concentration trends and habitat utilization. The peak and subsequent decline in mercury concentrations in whitefish may be evidence of a pulse of methyl mercury production in offshore areas of SIL caused by input of readily transported terrestrial organic debris immediately after flooding. The time course of mercury concentrations in whitefish has not followed trends in the abundance of benthic organisms in the SIL reservoir before and after lake impoundment. Total benthic biomass increased sharply between 1972, before impoundment, to 1977, one year after impoundment and remained quite constant at these elevated levels until at least 1983 (Wiens and Rosenberg 1984; Wiens and Rosenberg, unpubl. data). Unfortunately, there are no data concerning year-to-year differences in the methyl mercury content of SIL benthos. The pulse of methyl mercury production hypothesized for offshore areas soon after SIL impoundment could have resulted from the stimulatory effect on methylation rates of spruce needles and litter from the soil surface; spruce vegetation has been shown to be broken down rapidly when inundated (Crawford and Rosenberg 1984) and caused a stronger and more rapid enhancement of methyl mercury uptake in fish in limnocorrals than moss/peat material (Hecky et al. 1987). A longer term, but less severe enhancement of mercury methylation by moss/peat could be postulated and this longer term effect would be expected to lengthen the period of elevated mercury levels in fish species inhabiting the inshore zone where most of the organic matter added by flooding decays.

CONCLUSION

In the Churchill River Diversion area, fish in lakes which had their water levels raised have elevated mercury concentrations in their muscle. Increases have been directly observed at Southern Indian Lake, Issett Lake and Wuskwatim Lake and inferred by reference to nearby natural lakes, at lakes in and downstream of the Notigi Reservoir. Mercury concentrations in predatory fish are much higher in lakes in the Notigi Reservoir than in Southern Indian Lake and mercury concentrations in small fish show the same pattern (Bodaly et al. 1987a). Bodaly et al. (1987a) found no differences in mercury concentrations in flooded materials, especially soils to explain the differences in mercury concentrations in fish in the two reservoirs although flooded materials did have higher mercury concentrations than the natural lake sediments. The flooded materials did not have unusual mercury concentrations relative to the same materials in other regions. Bodaly et al. (1987a) and Hecky et al. (1987) could not find any simple, single factor relationship between common limnological measurements in limnocorrals or in impacted reservoirs to explain differences in mercury concentrations in fish among the reservoirs or over time within a reservoir.

Bodaly et al. (1987a) did find a significant relationship between mercury methylation rates at survey sites and mercury concentrations in small fish collected at those sites while Hecky et al. (1987) demonstrated that the addition of common flooded materials, soil moss-peat, spruce boughs and clay subsoil, did stimulate methyl mercury accumulation in small fish. Ramsey and

Ramlal (Appendices 3 and 4) have found that flooded boreal forest materials have higher mercury methylation rates than natural lake sediments while methyl mercury decomposition rates are similar to or often lower than the rates in natural lake sediments. The only reservoir tested which has recovered from a period of high mercury concentrations in fish, Cedar Lake, Manitoba, currently has very low methylation rates. However, marsh vegetation rather than boreal forest was flooded in this reservoir. The time course of recovery of Cedar Lake is likely inappropriate to reservoirs formed over boreal forest materials because Ramsey and Ramlal measured elevated mercury methylation rates in boreal forest reservoirs up to 40 year old.

Net methyl mercury production appears to be limited to a relatively short and specific time of the year in northern lakes and reservoirs. Maximum specific mercury methylation rates occur in late July through late August and are very low the remainder of the year, while specific rates of mercury demethylation do not show any strong seasonality. The inference from these patterns is that methyl mercury concentrations in the water and sediments will be highest in late July through late August. The maximum uptake of methyl mercury by fish in limnocorrals occurred in this same period of time with much lower uptake occurring in July and September. The same seasonal pattern of fish uptake of mercury occurred in control limnocorrals as well as in limnocorrals treated with flooded materials (Hecky et al. 1987), just as the same seasonal pattern of elevated specific rates of mercury methylation occurred in natural lakes as well as reservoirs. The addition of flooded materials to reservoirs has apparently amplified a natural cycle of net methyl mercury production. The implication is that the addition of flooded materials has given an advantage to those strains of bacteria which methylate mercury while having little or no effect on bacterial strains which demethylate. The general impression from the literature is that all bacteria can potentially methylate mercury as an incorrect synthesis when mercuric ion is present during normal growth, but there may be strains of especially efficient methylators which can gain a selective advantage by detoxifying their immediate environment by methylating mercuric ion (Wood et al. 1972).

It is not possible to say from the data available and techniques applied whether the peak in mercury methylating activity observed in reservoirs is due to a general enhancement of bacterial growth or the selection for efficient methylators in microenvironments where mercuric ion may be available at elevated concentrations. Further research on the microbial ecology of specific strains from the bacterial communities of rivers and lakes is necessary to understand this phenomenon. The same can be said of the mercury demethylating strains of bacteria. If a bacteria has the plasmid gene for demethylation, it likely will not be able to produce methyl mercury. However, the possession of the gene does not appear to confer any selective advantage within the lakes and reservoirs studied. It may be that methyl mercury concentrations do not become sufficiently elevated in these systems to become toxic to the microbial community. Again, greater knowledge of the ecology of the microbial community is required.

An annual pulse of net methyl mercury production even of short duration would be adequate to cause higher methyl mercury concentrations in fish. Fish rapidly take up and accumulate methyl mercury passively through their gills and especially through their diet if they are piscivorous. For example, Turner and Swick (1983) found the food pathway for mercury to be several times higher than uptake directly from the water for northern pike. The proportion of methyl mercury to total mercury in fish is relatively high, often greater than 90% in muscle, while the proportion in plankton and benthos is much lower

(Huckabee et al. 1979). Consequently, the dietary source of methyl mercury will be less important for benthivorous and planktivorous fish (Phillips et al. 1980). Fish can approach equilibrum with dissolved methyl mercury concentrations on the order of days to weeks, but they depurate methyl mercury from their bodies very slowly. For example, the biological half-life of methyl mercury in large fish is on the order of several months to a year. Consequently, a methyl mercury pulse of relatively short duration can cause methyl mercury concentrations in fish to rise and remain elevated long after the pulse has occurred.

There has been little change in the reservoirs of the Churchill River Diversion in most limnological parameters thought to be relevant to the elevated methyl mercury concentrations in fish muscle, since 1978 when full diversion flows began. Similarly, although mercury concentrations in fish have varied over time depending on species and region, there is now sufficient evidence to say that a rapid recovery to pre-impoundment levels of mercury concentrations in piscivorous fish has not occurred. Mercury concentrations in the piscivorous species are as high as they were in 1978-79 with little trend with time apparent on average. Although the record is shorter, there has not been a significant decline in specific mercury methylation rates since observations began in 1982. All these results are consistent with the hypotheses that net methylation rates have been more or less constant since 1978 and piscivorous fish mercury concentrations are in a quasi-steady state with regards to methyl mercury concentrations occurring seasonally in the reservoirs. The observed increase and subsequent decline to near pre-impoundment mercury concentrations in whitefish probably indicates that net methylation rates, at least in offshore areas, were higher immediately following impoundment than they are now. The occurrence of specific methylation rates in reservoirs up to 40 years old as high as the rates in the more recently flooded reservoirs suggests that the mercury concentrations in fish may remain elevated for a very long time. We would hypothesize that the mercury concentrations in fish will remain high until the organic materials stimulating mercury methylating bacteria are thoroughly decomposed or buried beneath sediment. The Southern Indian Lake reservoir is the first in which a complete record of mercury concentrations in fish and studies of the limnology of the reservoir are available extending from pre- through post-impoundment conditions. It is also the only one for which five continuous years of data on microbial specific mercury methylation rates are available. Therefore, we strongly recommend continued monitoring of these parameters to determine how long elevated mercury concentrations in fish persist and identify what environmental parameters are conducive to its recovery.

Mitigation of the mercury problem in the Churchill River Diversion area is probably not possible at present. Methyl mercury is entering the system diffusely over large areas of flooded terrain. In the long term, biological degradation of organics and mineral sedimentation will return the reservoirs to conditions comparable to natural lakes. In principle, any action which can accelerate these processes will improve conditions although acceleration may aggravate the problem in the short term, e.g. accelerating microbial degradation of organic substrates may increase mercury concentrations in fish in the short term. As a preventive measure, removal of vegetation and organic soil prior to impoundment should reduce if not eliminate the increase in mercury concentration in fish after impoundment. Such a measure may be prohibitively costly in large reservoirs before impoundment, and almost certainly is after impoundment.

Only one specific mitigation measure, sediment resuspension, was subjected to experimental test. The English-Wabigoon study (Canada-Ontario Steering Committee 1983) recommended semi-continuous resuspension of clean (low mercury content) lake sediment be explored as a pilot project to determine its efficacy in reducing bioavailability of mercury in the water column and in surficial sediments. In our limnocorral experiments, neither modern lake sediments of low mercury content nor eroded glacial Lake Agassiz sediments of low mercury content reduced bioavailability of mercury in the short term of the experiment. Comparison of Southern Indian Lake where impoundment increased turbidity in the water column with Notigi Reservoir where impoundment reduced turbidity has provided a whole lake experiment to test the efficacy of sediment resuspension. Concentrations of mercury in fish from Southern Indian Lake are lower than in Notigi, but not lower than would be expected from the lesser amount of flooding in Southern Indian Lake (Bodaly et al. 1984a). In fact, within Southern Indian Lake there is not a substantial difference in mercury concentrations in small or large fish between turbid regions of the lake such as region 6 and clear water areas such as region 5. Therefore, we cannot recommend sediment resuspension as a mitigation technique.

The English-Wabigoon study examined selenium treatments as a mitigatory action. Selenium was found to be less effective than resuspended sediments but it did reduce mercury uptake into biota at concentrations of 1 $\mu g \cdot L^{-1}$, well below drinking water standard concentrations. However, selenium is toxic to many aquatic organisms, as well as man, and it was recommended that extensive further studies including whole lake experimental treatments under controlled conditions would be required before any application could be recommended. Treatment control would be much more difficult in large diffuse source systems like Southern Indian Lake and Notigi Reservoir in contrast to industrial point sources of mercury where selenium could be economically added at the source of mercury.

A possible solution may lie within the cause of the problem, the microbial balance between methylation and demethylation. If our knowledge of the species ecology were better, it might be possible to selectively add substances which would enhance demethylation or inhibit methylation. Although the biochemical pathways for these reactions are reasonably well-studied and understood, the distribution of the biochemical systems within the species in natural bacterial communities and the adaptive role of the biochemical pathways within the general ecology of the host species is not known. We have identified a mercury demethylation potential within all the lakes and reservoirs studied, but this potential does not seem responsive to reservoir formation in contrast to the mercury methylating bacteria. Genetic engineering, altering the microbial pathways, may be possible, but this could not be done without understanding the adaptive roles of these reactions in their host species and their significance to other biogeochemical cycles. Greater knowledge of species ecology of the microbial communities is required before any further progress can be made in preventing or mitigating the occurrence of elevated mercury concentrations in fish in new reservoirs.

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Table 1. Changes in morphometric relations in the Southern Indian Lake and Notigi Reservoirs, from Newbury et al. (1984). Pre-impoundment conditions for Notigi Reservoir are aggregate totals for the seven small lakes amalgamated by impoundment.

	Southern Indian Lake		Notigi Reservoir	
	Pre	Post	Pre	Post
Total water area (km^2)	1977	2391	153	584
Flooded area (km^2)	0	414	0	431
Volume (km^3)	16.8	23.4	0.49	4.1
Mean depth (m)	8.5	9.8	3.2	7.8
Volume ÷ flooded area (m)	0	56.5	0	9.5
Water renewal (d)	190	260	184	62

APPENDIX 1

LIMNOLOGICAL DATA FOR THE SOUTHERN INDIAN LAKE AND NOTIGI RESERVOIRS

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Table 1. Physical and chemical conditions in Southern Indian Lake Reservoir and Notigi Reservoir, August 18-20, 1983.

Location	Temperature	conductivity	Secchi	Oxygen	Oxygen
	°C	µS·cm⁻¹	m	mg·L⁻¹	% sat.
Churchill River	19.9	65		9.0	90
Southern Indian Lake					
Region 1	17.9	70	0.6	8.0	90
Region 2	17.6	83	0.8	8.4	94
Region 4	17.0	127	2.2	8.6	95
Region 5	16.7	110	2.6	8.6	94
Region 6	17.0	88	0.7	8.4	93
Notigi Reservoir					
Notigi West	18.0	91	1.5	7.1	80
Notigi East	18.6	94	1.5	7.5	86
Issett	17.6	112	1.5	7.0	79
West Mynarski	18.3	132	1.3	7.1	81

Temperature at 1m depth for Station 4B-SIL: 1972-1985.

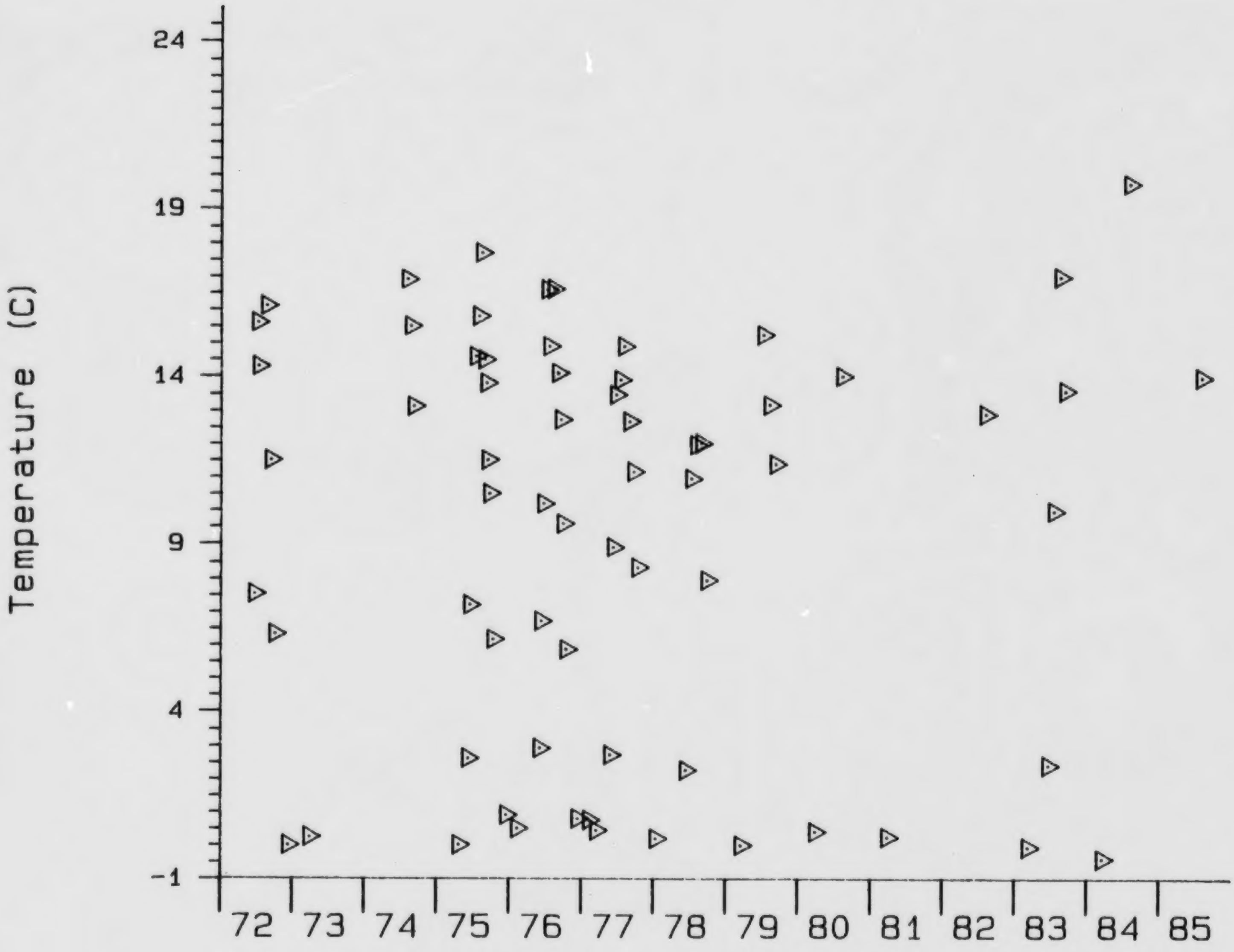


Fig. 1. Temperature at 1 m depth in region 4 on Southern Indian Lake: 1972-1985.

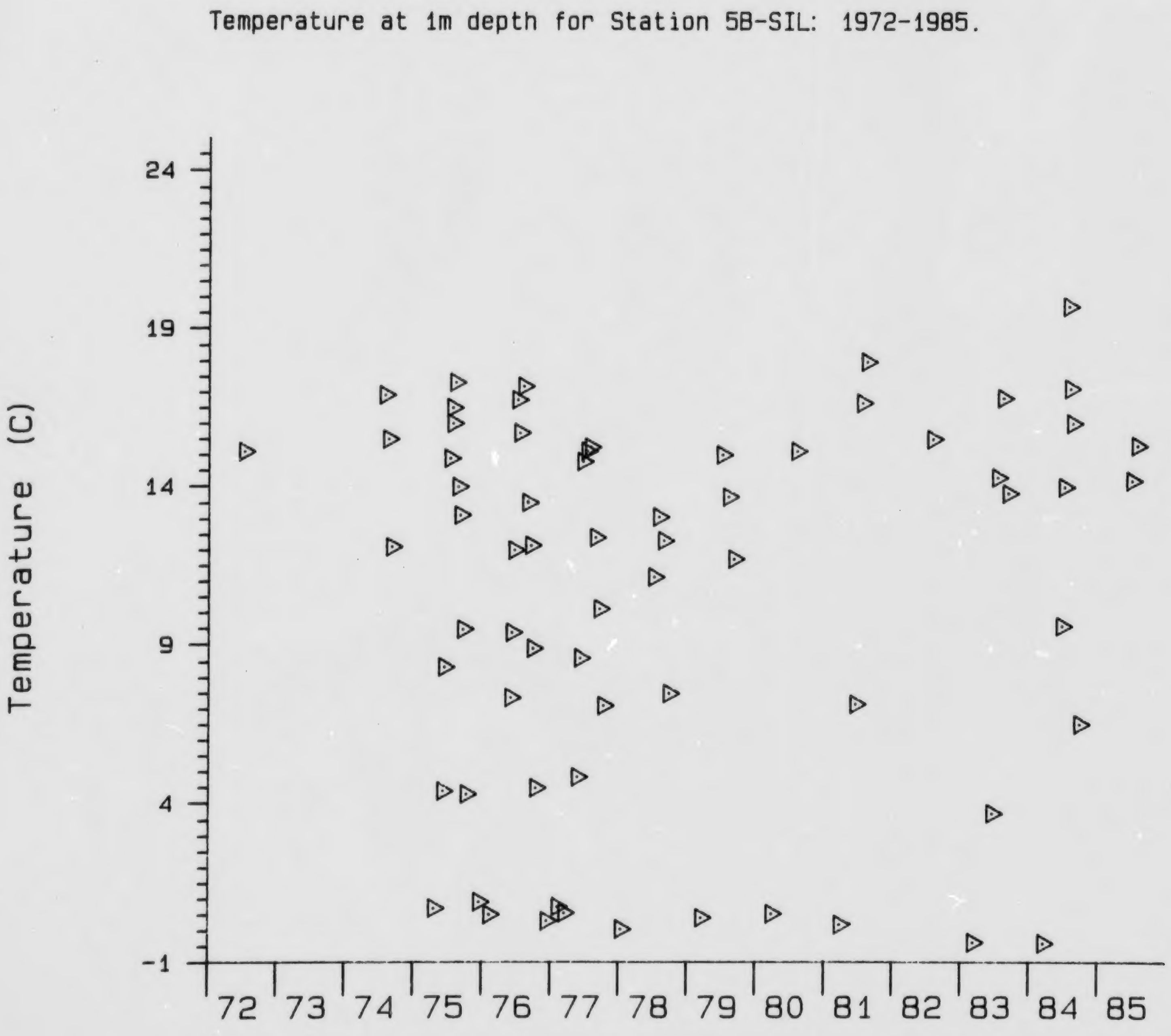


Fig. 2. Temperature at 1 m depth in region 5 on Southern Indian Lake: 1972-1985.

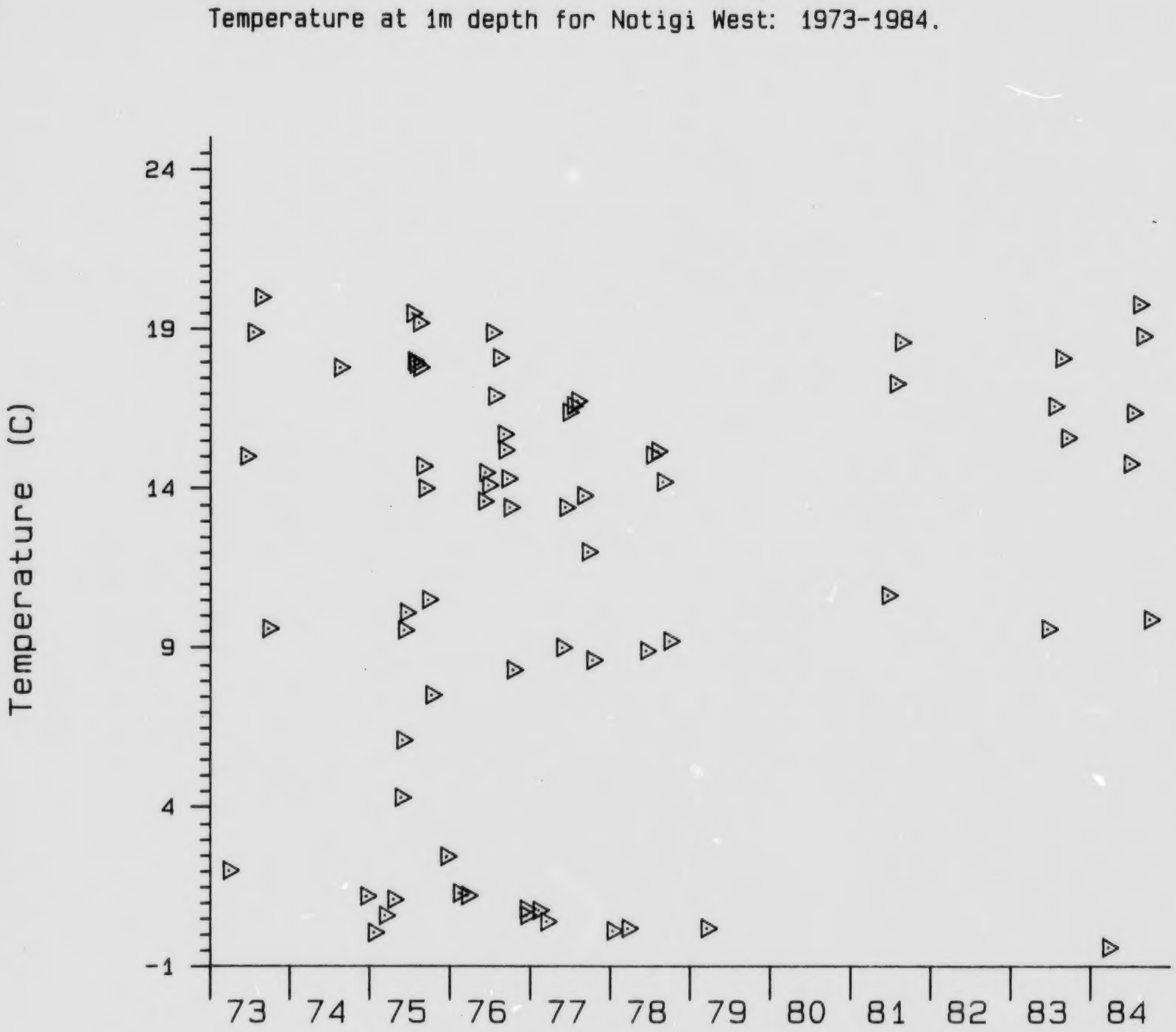


Fig. 3. Temperature at 1 m depth in the west basin of Notigi Reservoir: 1973-1984.

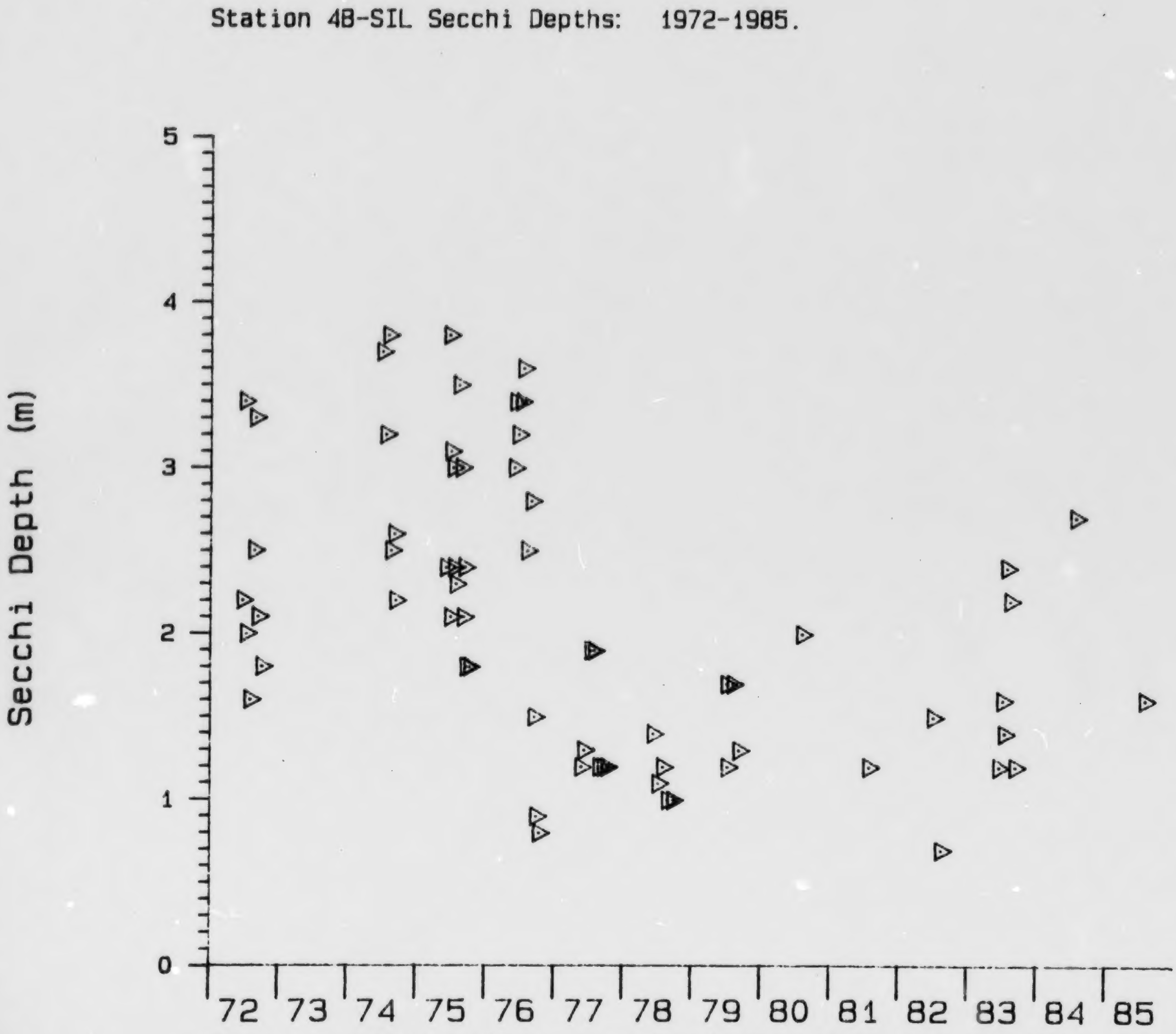


Fig. 4. Secchi disk depths in region 4 on Southern Indian Lake: 1972-1985.

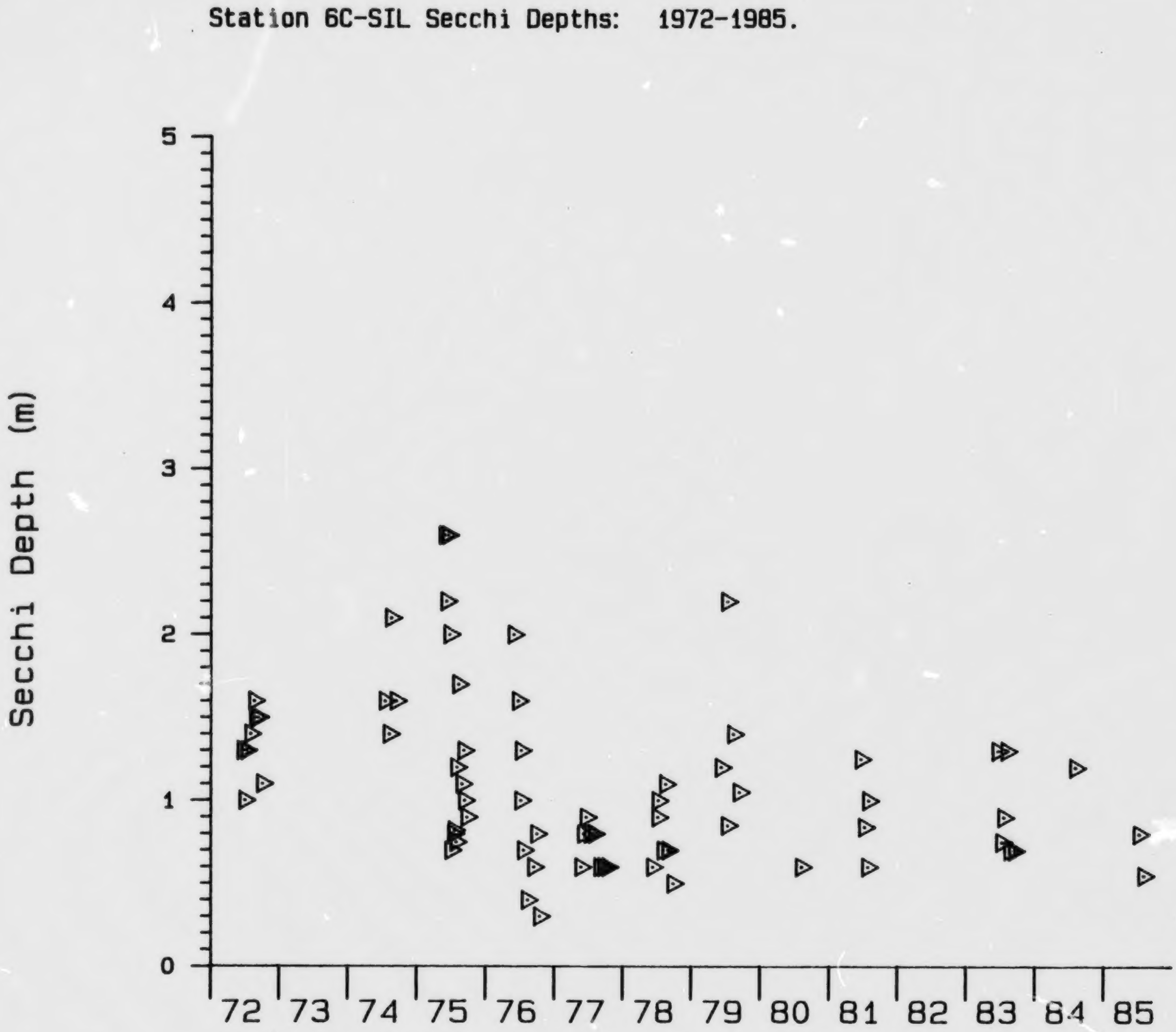
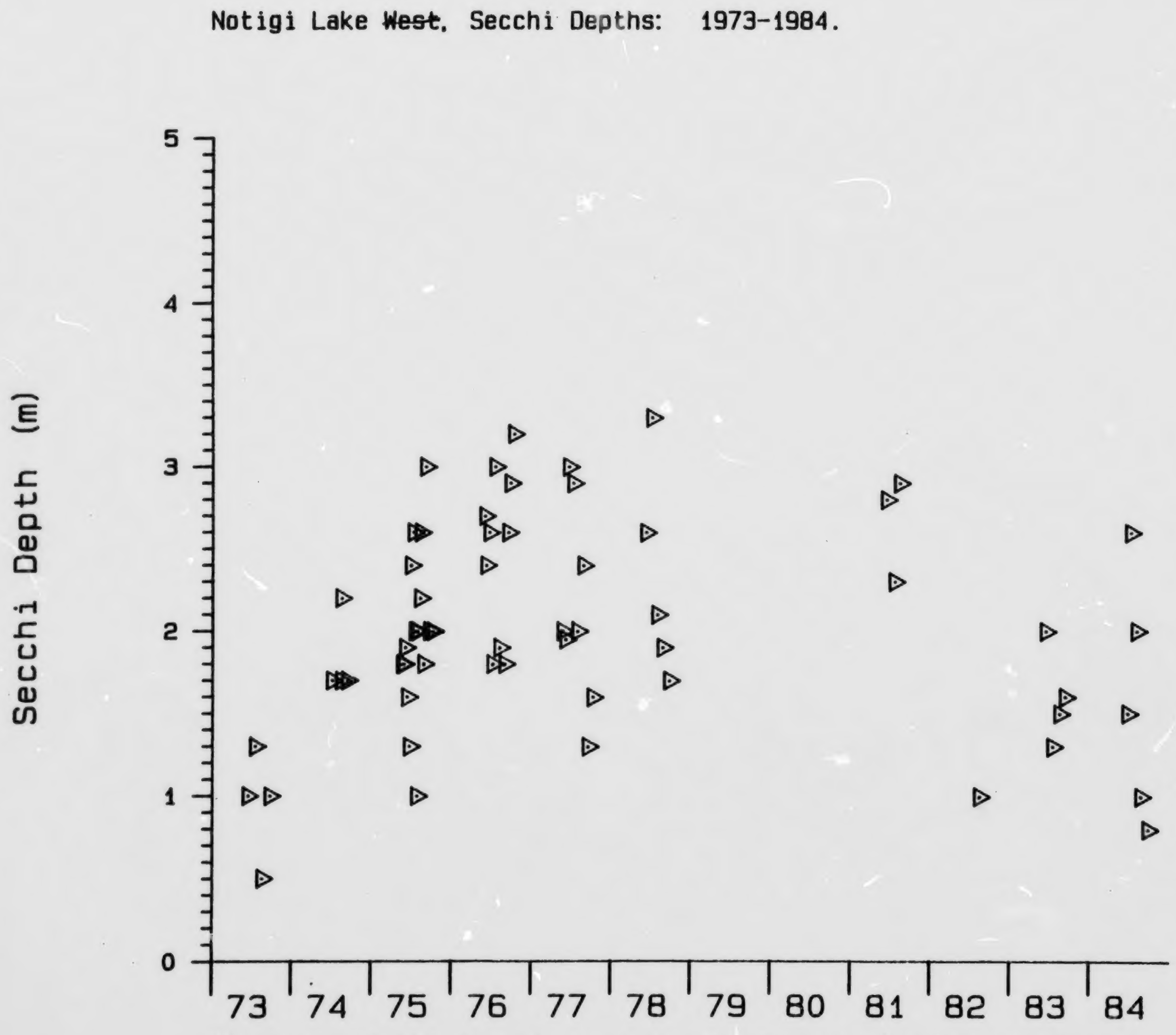


Fig. 5. Secchi disk depths in region 6 on Southern Indian Lake: 1972-1985.



Notigi Lake ~~West~~. Secchi Depths: 1973-1984.

Fig. 6. Secchi disk depths in the west basin of Notigi Reservoir: 1973-1984.

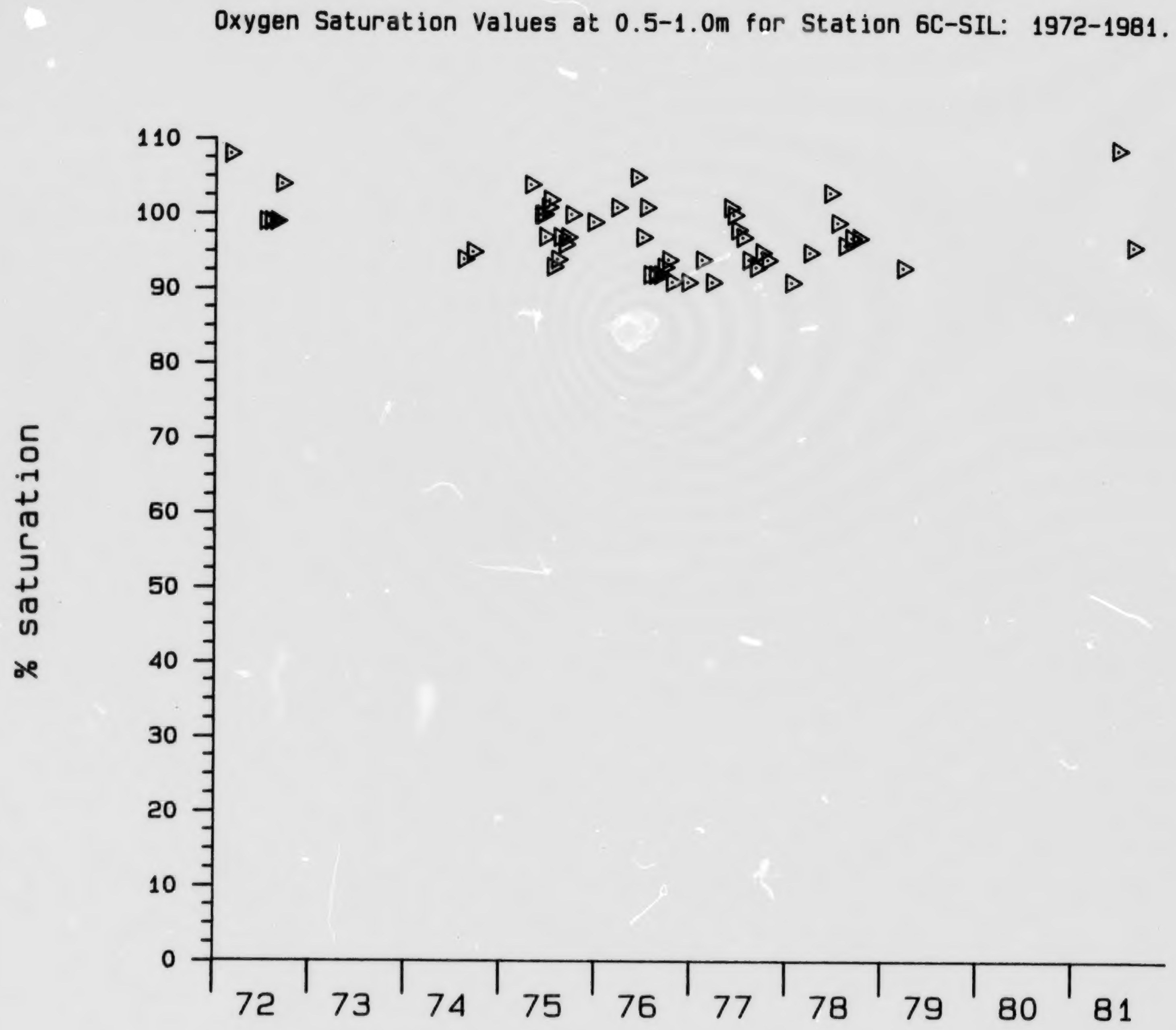
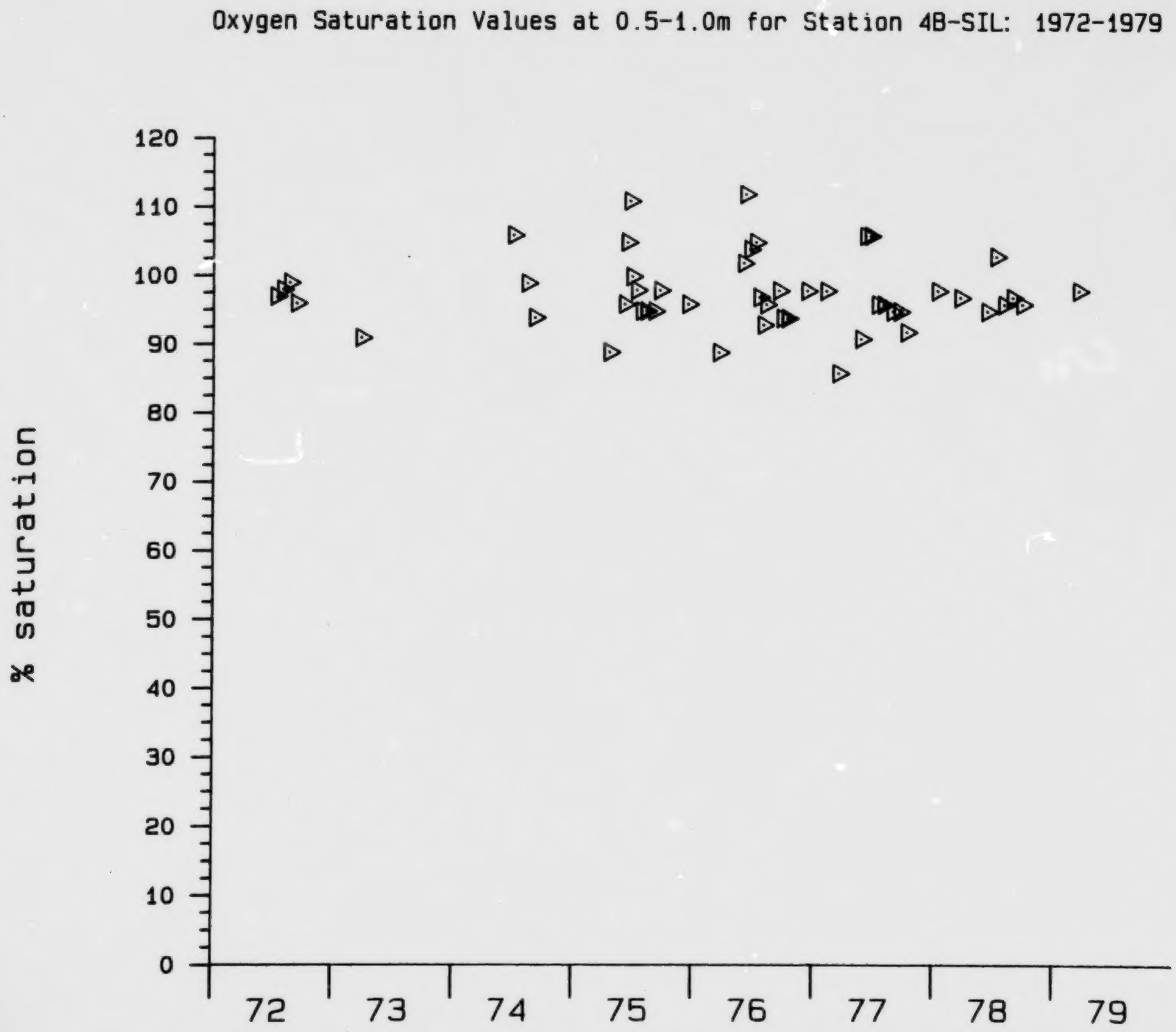
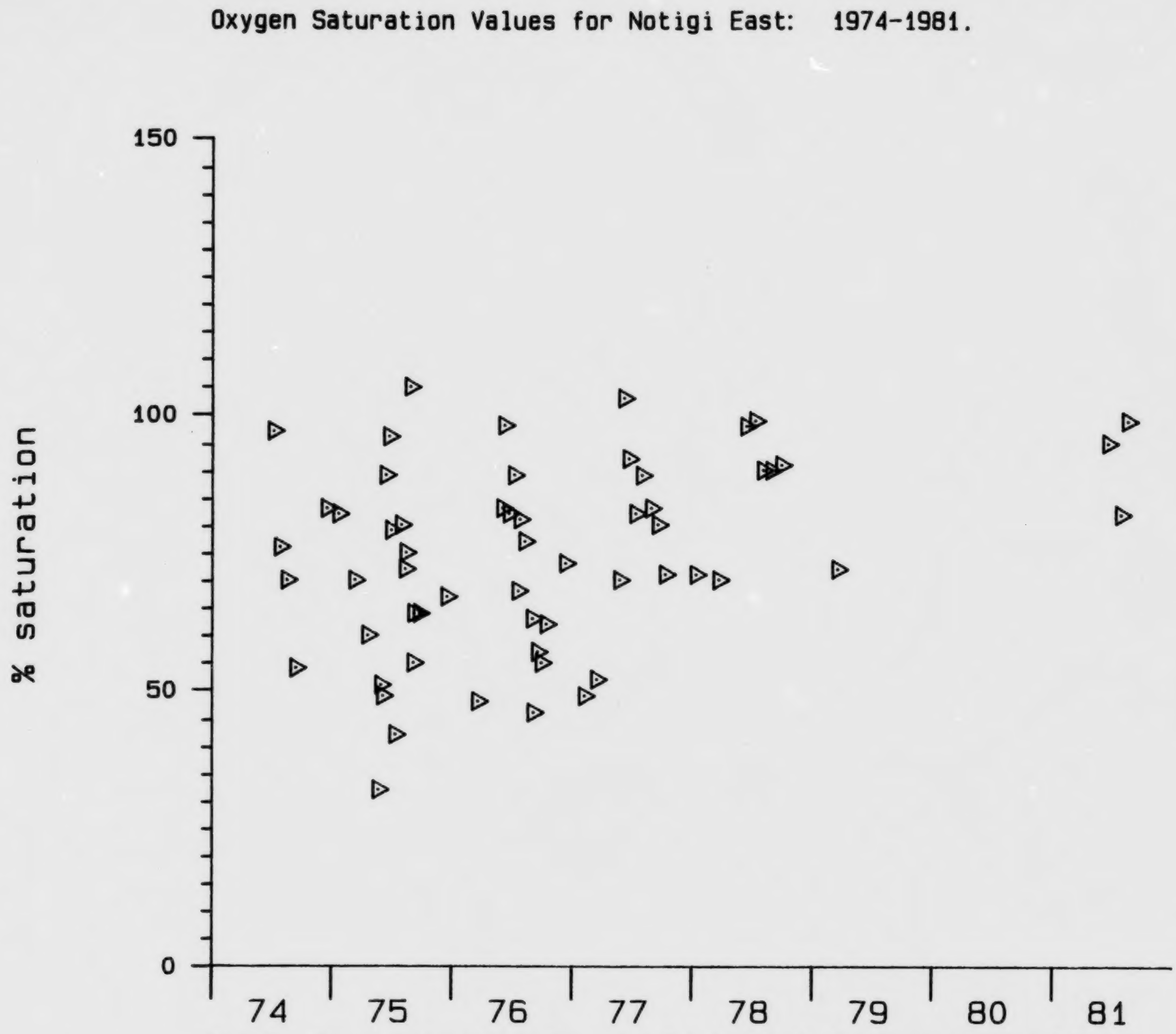


Fig. 7. Oxygen saturation values at 0.5-1.0 m depth in region 6 on Southern Indian Lake: 1972-1981.



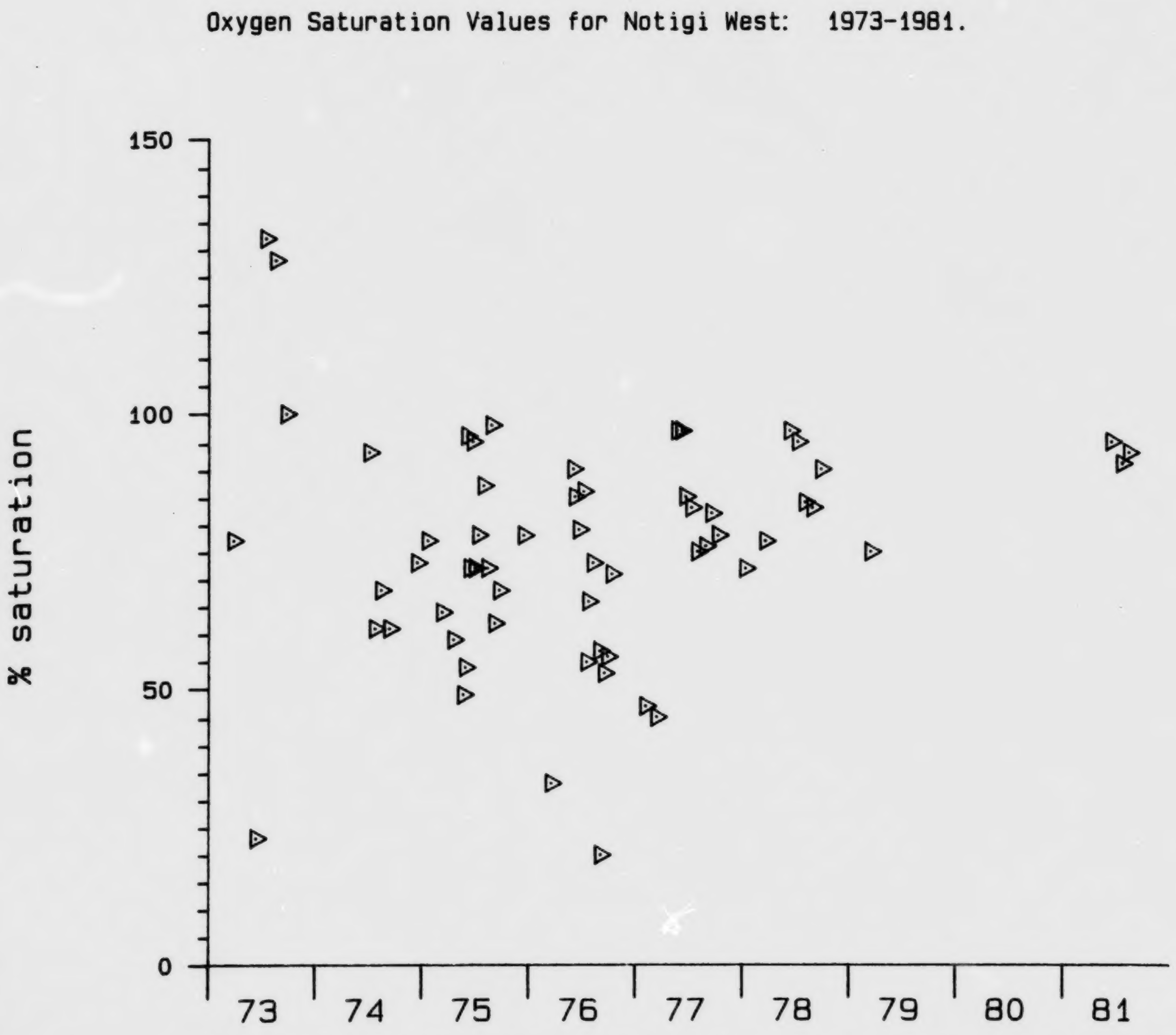
Oxygen Saturation Values at 0.5-1.0m for Station 4B-SIL: 1972-1979

Fig. 8. Oxygen saturation values at 0.5-1.0 m depth in region 4 on Southern Indian Lake: 1972-1979.



Oxygen Saturation Values for Notigi East: 1974-1981.

Fig. 9. Oxygen saturation values at 0.5-1.0 m depth in the east basin of Notigi Reservoir: 1974-1981.



Oxygen Saturation Values for Notigi West: 1973-1981.

Fig. 10. Oxygen saturation values at 0.5-1.0 m depth in the west basin of Notigi Reservoir 1973-1981.

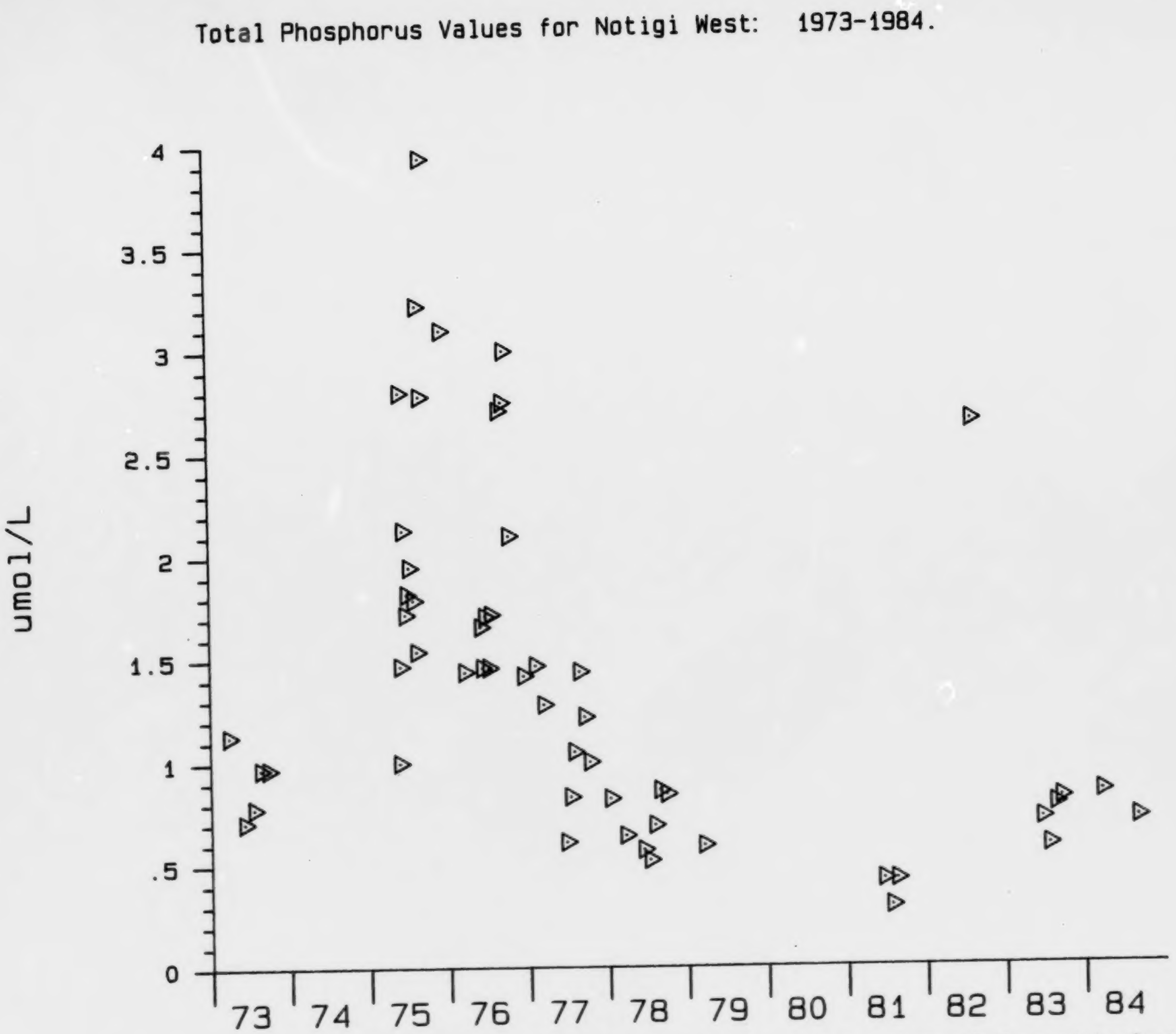


Fig. 11. Total phosphorus concentrations in the surface waters of the west basin of Notigi Reservoir: 1973-1984.

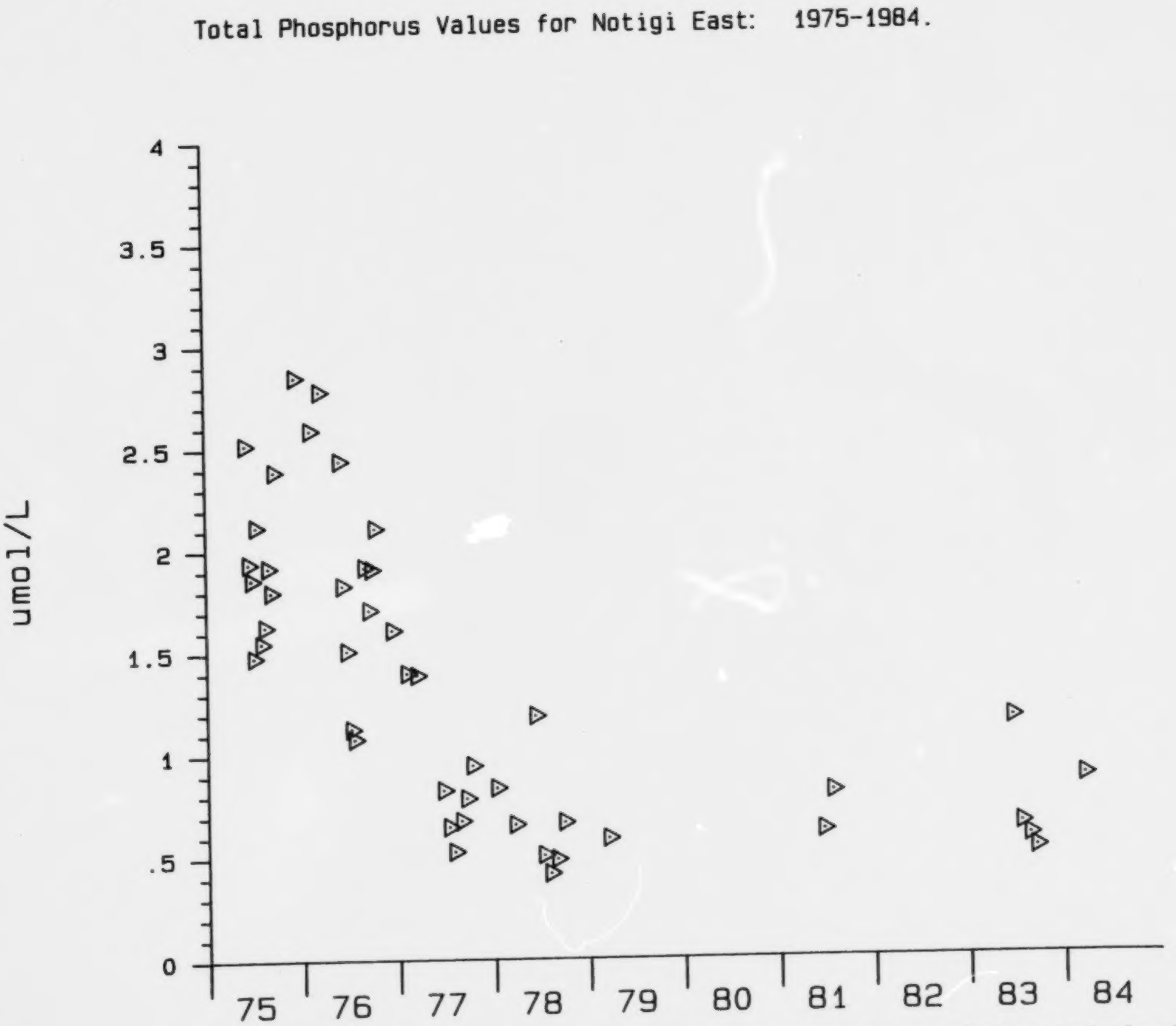


Fig. 12. Total phosphorus concentrations in the surface waters of the east basin of Notigi Reservoir: 1975-1984.

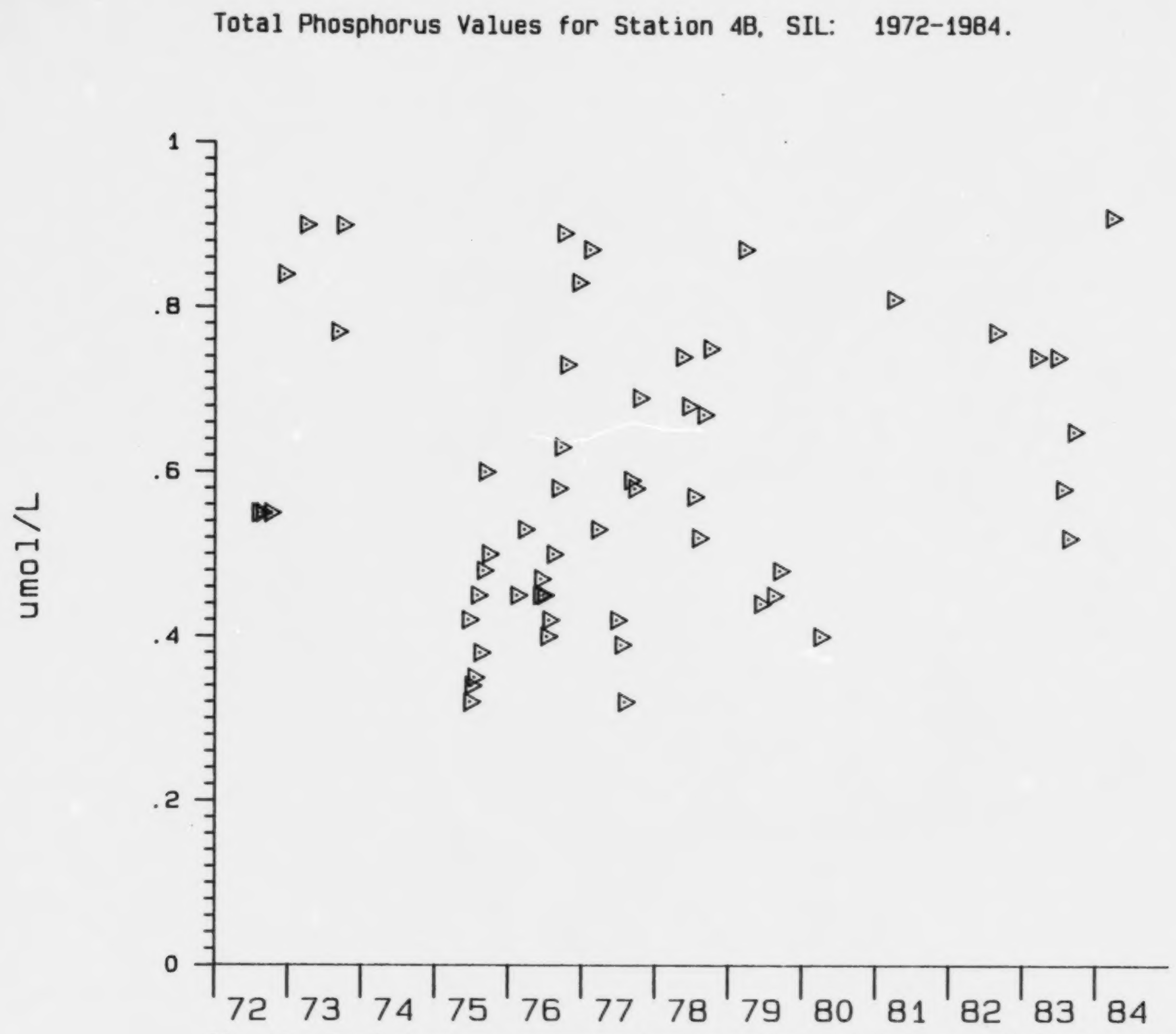


Fig. 13. Total phosphorus concentrations in region 4 of Southern Indian Lake: 1972-1984.

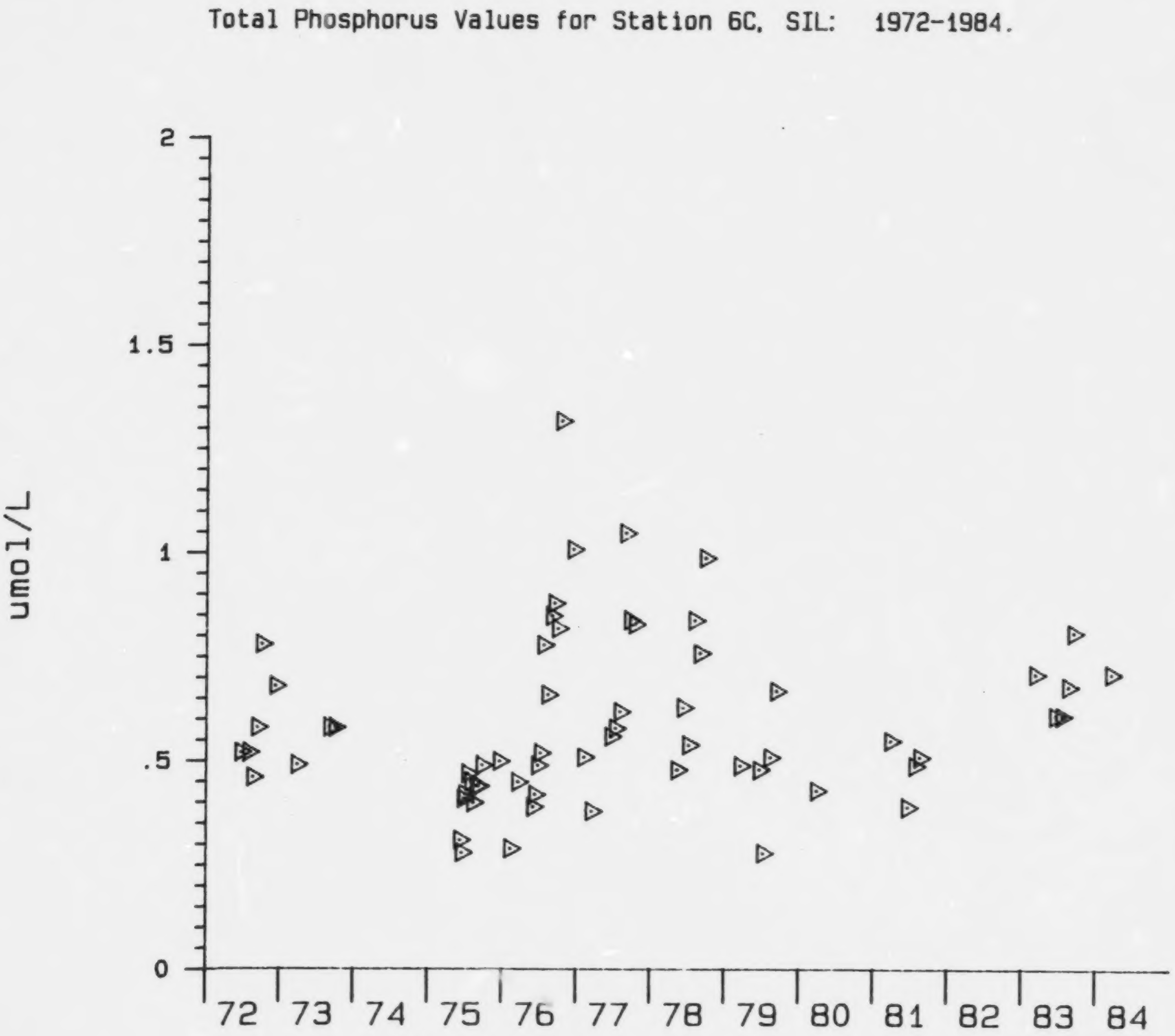


Fig. 14. Total phosphorus concentrations in region 6 of Southern Indian Lake: 1972-1984.

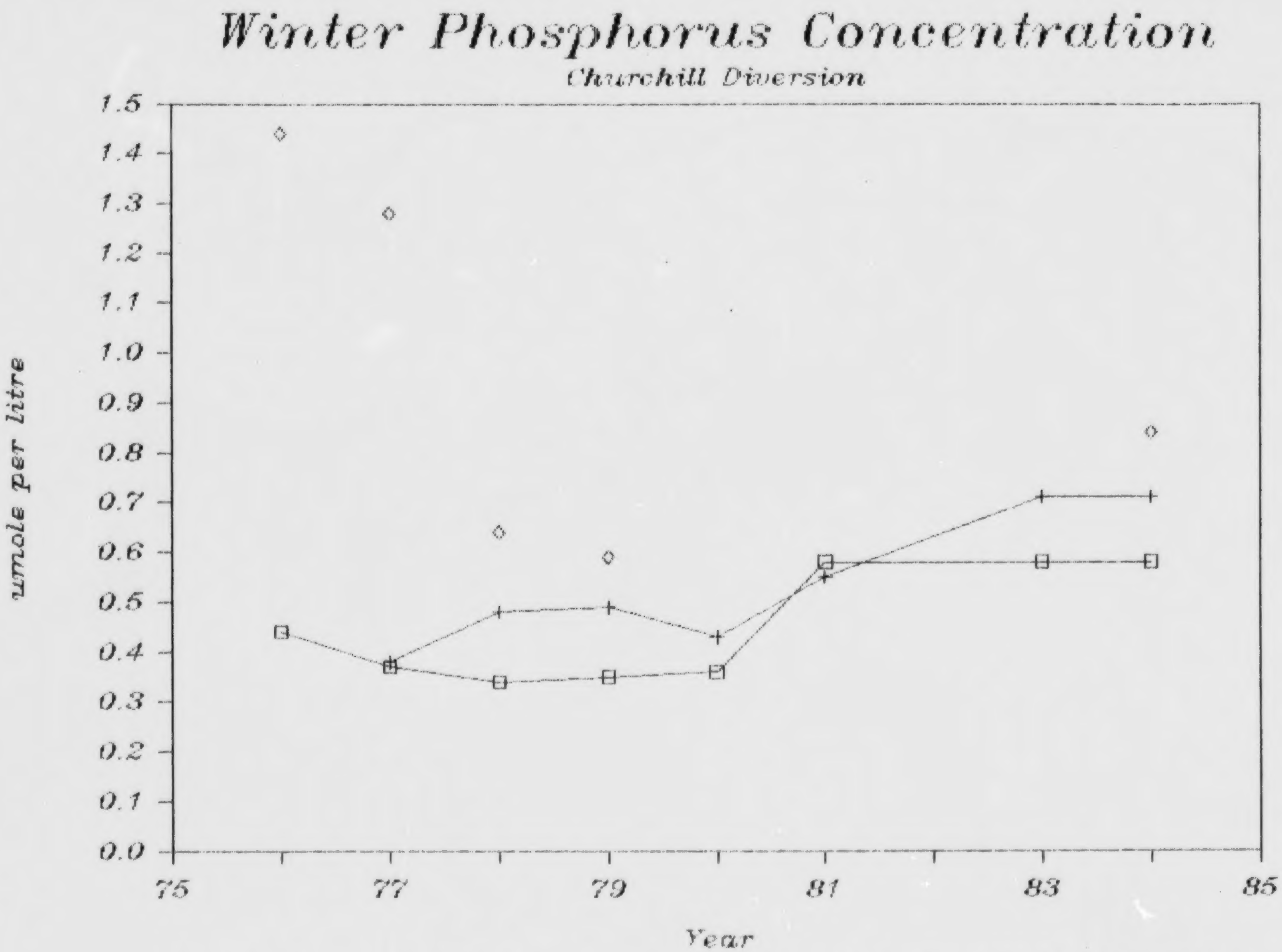


Fig. 15. Comparison of March phosphorus concentrations in the Churchill River (⊟—⊟), diversion outflow from Southern Indian Lake (+ —— +), and outflow from Notigi Reservoir (◇) from the year of impoundment and initial diversion through 1984.

SPECIAL NOTE

This report was prepared for the Canada-Manitoba Agreement on the Study and Monitoring of Mercury in the Churchill River Diversion. Any conclusions, recommendations, or opinions expressed herein are those of the author and are not necessarily the same as those of the Steering Committee for this Agreement.

APPENDIX 2

SHORELINE EVOLUTION IN THE SOUTHERN INDIAN LAKE RESERVOIR

by

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INTRODUCTION

The raising of Southern Indian Lake by 3 m in 1976, to facilitate diversion of the Churchill River to hydro-electric stations on the Nelson River, caused severe shoreline erosion. The results of a study of that erosion are reported in Newbury and McCullough (1984). Erosion in the early years of impoundment was as high as 20 m^3 per m of shoreline per year and was estimated to remove about 4000 tonnes of mineral materials from the whole shoreline per year, or more than an order of magnitude greater than the annual sediment load of the inflowing Churchill River. Hecky and McCullough (1984) reported that 90% of this material remained in Southern Indian Lake, ultimately as bottom sedimentary deposits. On the basis of cores of the bottom deposits near two erosion monitoring sites, it was suggested that much of this material was deposited in a narrow zone near the eroding shores, in water as shallow as 2 m depth. It was hypothesized that these deposits would be unstable in the long term. Their erosion would continue after erosion of the actual bank had declined or ceased, causing unnaturally high lake turbidity to persist after the shorelines had become more generally stable.

In this report, preliminary results of a study of nearshore sedimentation processes at Southern Indian Lake are described. It is shown that the ratio of the weight of the nearshore sedimentary deposit to the weight of eroded bank material ($R_{d:e}$) is generally lower where beach development or exposure of bedrock has begun to affect the erosion rate. An inverse relationship between $R_{d:e}$ and incident wave energy is apparent in the data.

METHODS

Site Selection

In 1975, 20 sites were selected for monitoring of shoreline erosion. Until 1981,tape and level surveys of bank profiles were made annually at each of these sites (Newbury and McCullough 1984). In 1983, several sites were chosen from among these, and additional sites were selected, for measurement of the total volume of material eroded from the shoreline, and of that deposited near the shore, since impoundment of the lake in 1976. Locations of sites discussed in this report are identified in Fig. 1. (All shoreline monitoring sites previously described by Newbury and McCullough retain the original numbers and new sites are numbered from 21). The major selection criteria were as follows:

1. Easily determined volume of postimpoundment nearshore deposits: Previous experience had shown that postimpoundment sediments could be relatively easily distinguished from preimpoundment sediments where clay banks were eroding at sites of preimpoundment sand or gravel beaches, or

in regions where there had been gently sloping bedrock near the preimpoundment shoreline.

2. Range of fetches: It was desired to use this erosion data to test the value of K_e, the erosion index determined by Newbury and McCullough (1984), and to test for a possible relationship betweeen $R_{d:e}$ and incident wave energy. Sites were chosen with a range of fetches from 2 to 30 km to ensure that data would be drawn from a sample with a wide range of incident wave energy.
3. Eroding clay banks, with and without the presence of sand: It was observed in earlier erosion studies that the disaggregation of the eroding clay appeared more complete where sand was present at the water's edge. Extensive nearshore deposition is believed to occur because much clay is not broken down to primary particle size, but instead is carried through the surf zone and deposited as sand to pebble-size aggregates. Thus sand, acting as an abrasive, may increase the proportion of eroded clay abraded sufficiently to be carried far offshore in suspension.
4. Shorelines reaches beginning to be stabilized by development of sand or gravel beaches or by exposure of bedrock at the water's edge: It was hypothesized that nearshore deposits are a dynamic feature, and will themselves erode when the bank erosion rate declines enough that the rate of supply of sediment to the nearshore zone is exceeded by the offshore sediment transport rate. Hence, $R_{d:e}$ was expected to be lower at more stable shoreline sites.

Shoreline Erosion Measurement

Airquest Surveys Limited was contracted by the Department of Fisheries and Oceans to take aerial photographs and plot shoreline maps and bank profiles at selected sites. In July, 1983, vertical aerial photographs were taken at a scale of 1:10,000. Maps and profiles were photogrammetrically plotted using these and existing 1970 aerial photographs at a scale of 1:30,000. (See Fig. 2 for maps of profiles and nearshore transects for sites described in this report). Seven to 16 profiles were drawn at each site equidistantly spaced to sample a reach of 0.5 to 1 km. The density of profiles drawn for each site was roughly proportional to the irregularity of the shoreline (Fig. 2). Along irregular stretches of shoreline, reaches were chosen to include a representative sample of local headlands and indentations. The displacement between each 1970-1983 profile pair was the measure of the erosion of the bank since impoundment. The weight of sediment eroded was calculated from volume as described by Newbury and McCullough (1984), and assuming the same ice content (58% of dry weight) of the bank materials.

Nearshore Deposition Measurement

Postimpoundment nearshore deposits were measured by coring and by probing with a survey rod. At each site, one transect perpendicular to the shoreline was traversed by a SCUBA diver, and the bottom surface was described from the water's edge to 300 m offshore. Distance was measured with a marked rope tied near the water's edge. At selected distances, sediment cores were taken with 63.5 mm diameter plastic tubes. Cores were examined and described in the field, with emphasis on determining the postimpoundment-preimpoundment discontinuity.

From a sample of cores of postimpoundment nearshore sediments (in waters of less than 6 m depth) at several sites, a mean dry density of 972 kg m^{-3} (s=78, n=11) was determined. Weights of postimpoundment deposits reported below were calculated from volume using this figure.

In addition, at each site, 3 to 4 transects perpendicular to the shoreline were investigated by probing from a boat with a survey rod, distance from the water's edge being measured with a survey tape tied at the water's edge. Where the shoreline was particularly irregular, transects were chosen to be representative of the distribution of headlands and indentations (Fig. 2).

Only data from survey rod probe measurements is presented in this report. In general postimpoundment clay deposits were easily penetrated by the survey rod. Where the preimpoundment surface was of coarse materials (sand, gravel, cobbles) or bedrock, the interface was easily felt. Also, an erosional surface on overconsolidated glaciolacustrine clay (bank material before impoundment) was easily distinguishable from a recent lacustrine clay deposit by the difference in penetrability. Postimpoundment sediment thickness was determined at short (1 to 3 m) intervals from the water's edge out to 3 to 5 m depth along each transect. Areally coincident core and probe data were found to reproduce each other within 2 to 3 cm. As a large fraction of the nearshore deposits mapped occurred in beds of 0.3 to 0.8 m thickness, and as error in offshore distance measurements over 100 m was minor, the accuracy of the method can be estimated to be better than 10% for each transect.

Wave Energy Calculation

Wind data has been recorded at the Fisheries Research Station near the community of Southern Indian Lake since 1976. The hourly wind record, from a 10 m high anemometer (see Fig. 1 for location), was altered by overwater: overland ratios in accordance with Richards and Phillips (1970). The duration of the wind was compiled by speed class and direction for the open water period from September, 1976 (month of impoundment) to July, 1983. Wave energy was hindcast for each wind class for each study site (USCE 1977), and multiplied by wind duration to determine total wave energy incident on the shoreline during the period of erosion. The method is described in Newbury and McCullough (1984).

RESULTS AND DISCUSSION

Shoreline erosion

Photogrammetric profile data varied in quality depending on site conditions. As a simple test of vertical resolution, the backshore, uneroded zone could be expected to coincide on superimposed 1970 and 1983 profiles. Good coincidence (within 0.2 to 0.3 m) was found at sites with a backshore cover of treed muskeg (open, small black spruce); profiles at sites with closed tree cover often failed this test. Better results were obtained at sites where a flat or gently-sloping backshore was intersected by a steep erosion scarp than at sites where the break in slope at the bank was more subtle. Most of the latter sites are excluded from the following discussion.

Measured volumes of shoreline erosion and calculated weights of eroded sediment are listed for 21 sites in Table 1. For each site, the number of profiles used to determine the mean erosion is indicated, together with the standard deviation of the data. Although there is considerable imprecision inherent in the aerial photogrammetric method at the scale of available photography, much of the high standard deviation may be ascribed to real differences in erosion from line to line along the shoreline. Local longshore variance in erosion is particularly high along irregular shorelines, where headlands were cut away at a much higher rate than protected bays and indentations.

The erosion index, (Ke) calculated by dividing the volume of shoreline erosion by the total incident wave energy for the period of erosion, is reported in Table 2. Sites were divided into 2 classes: sites with only clay and silty-clay exposed in the bank and on the foreshore; and sites where a beach of coarse materials had developed or considerable bedrock had been exposed at the water's edge along the shoreline reach. Sites where this distinction could not be made unambiguously are not included in Table 2. At all measured sites with coarse sediments or bedrock at the foreshore, the eroding bank had been predominantly of clay; in most cases coarse beach material at these sites was imported from local sources (e.g. esker and proglacial deposits intersecting the shoreline) by littoral drift.

The mean erosion index for clay shorelines, K_e=0.00038, is very similar to K_e=0.00035 reported by Newbury and McCullough (1984) for the early years of impoundment. The rate of erosion in clay shorelines has not been impeded by foreshore development.

On the other hand, at sites where considerable beach development (sites 11, 56, 57, 62, 63) or bedrock exposure (sites 11, 46) had occurred at the date of survey, the mean K_e=0.00005. At all of these sites, immediately after impoundment, the shoreline was comprised of glaciolacustrine sediments or of clayey till. Previous investigations of shoreline erosion showed similar, higher K_e for these materials. For example, some bedrock was observed at the water's edge at site 11 as early as 1978; in 1977, K_e=0.00054 was reported. The long term K_e=0.00004 for site 11 is indicative of neither the initial, higher rate of erosion of clay overburden, nor the rate in 1983, when the backshore was protected partly by bedrock and partly by beach; it is simply a cumulative mean for the period. For all of the sites showing either beach development or bedrock exposure, the low K_e

may be taken as evidence of the effectiveness of shoreline stabilization processes in decreasing the rate of bank erosion.

Nearshore Deposition

Volumes of postimpoundment nearshore deposits measured at each site are reported in Table 3. Data reported is the mean volume from all transects measured, from the water's edge to 100 m offshore. Distance was chosen over depth as a defining characteristic of the nearshore zone even though it might be expected that any relationship between nearshore deposits and incident wave energy would be dependent on depth rather than on distance from shore. Unfortunately, small differences in the offshore slope have a large effect on the distance from the water level to any given depth. For the sites listed in Table 3, the distance from the water's edge to the 4 m isobath ranges from 17 to 108 m. Thus, if the extent of the nearshore zone were defined by depth, variance (in the length of the cross-section and, consequently, in the calculated volume of the deposit) due to slope would predominate over any other possible parameters.

The high standard deviation among transects is a product of real variance. Sediments tend to be swept away from headlands and towards indentations in the shoreline. Local irregularities in the bottom topography, usually due to local bedrock high points, also affect sediment transport.

Ratios of the weight of the nearshore deposit to the weight of sediments eroded from the adjacent bank ($R_{d:e}$) are listed in Table 4. At unstabilized eroding clay shorelines, 0.54 (range: 0.28 to 0.78) of eroded sediments were found deposited within 100 m of the water's edge. Major nearshore postimpoundment deposits typically formed narrow lenses, thickening quickly from the shoreward apex to a maximum at 10 to 50 m from shore, and tapering gradually to a few centimetres thickness by 100 to 150 m offshore. The mean maximum thickness of measured nearshore deposits was 0.7 m (range: 0.2 to 1.2 m, 17 sites) at eroding clay banks. At the same sites, the maximum thickness occurred at a mean depth of 2.2 m (range: 1.3 to 3.9 m). Inshore of these deposits, along unstabilized clay shorelines, an erosional shelf had formed on indurated, glacio-lacustrine clay.

Figure 3 shows an inverse relationship between annual incident wave energy and $R_{d:e}$ at unstable clay shorelines (solid circles). The general trend of the data implies that for shorelines with an annual incident wave energy >120,000 tonne-m per m most of the eroded material would be carried beyond 100 m from the shore.

Mean $R_{d:e}$ for shorelines affected to some degree by beach formation or bedrock exposure is only 0.13 (range: 0 to 0.27). Based on the general trend described above, high incident wave energies at sites 11 and 57 would be expected to remove most eroded bank sediments far offshore even at clay shorelines. However, for sites 56, 62, and 63, within the same range of wave energy as the data for eroding clay shorelines, the mean $R_{d:e}=0.20$ and individual values plot below the general trend for eroding clay banks (Fig. 3, solid triangles). For these sites, a smaller fraction of the eroded bank sediments were found deposited nearshore than is implied by the general trend for eroding clay shorelines.

It was a goal of the investigation of nearshore erosion to examine the effect on $R_{d:e}$ of: 1) the presence of sand acting as an abrasive on clay aggregates (hypothesis 3, above), and 2) decreased erosion rate due to shoreline stabilization (hypothesis 4). However, at sites with considerable sand (sites 56, 57, 62, and 63) beaches of coarse sediments were already apparently effective in slowing the erosion rate (Table 2). At site 11, where exposure of bedrock at the water's edge caused the decline in the shoreline erosion rate, sand beaches were forming in small bays, indicating a significant presence of sand. At all of these sites, by 1983, it was no longer possible to separate the effect of sand from the effect of the declining erosion rate on $R_{d:e}$.

Some evidence from 2 other sites where much bedrock has been exposed, but where negligible sand or gravel was found along the shoreline, suggests that sand may be important. At site 46, considerable stretches of bedrock had been exposed at the water's edge by 1983. Data was not included in Table 4 because the nearshore deposits were only 50 m from shore. However, within 50 m, the volume of postimpoundment sediments was 10 m^3 (9800 kg) per m of shoreline; as the volume eroded was 21 m^3 (21800 kg) per m of shoreline, for deposits within 100 m of shore $R_{d:e}>0.45$. At site 29, also with exposed bedrock in 1983, the profile data was adequate only to conclude that the volume of bank erosion was <20 m^3 (20800 kg) per m of shoreline. Deposition within 35 m of the water's edge was 9 m^3 (8700 kg) per m of shoreline; hence, $R_{d:e}>0.42$. Although there was visual evidence of shoreline stabilization at both sites by 1983, $R_{d:e}$ would nonetheless have been within the range expected for eroding clay banks. At least at moderate energy-level sites like sites 29 and and 46, any decline in bank erosion rates had not yet resulted in a significant loss of nearshore deposits. That there is a smaller $R_{d:e}$ where beaches have formed along stabilizing shorelines suggests that the presence of sand is an independent factor in determining $R_{d:e}$.

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Table 1. Mineral sediment eroded at selected shoreline sites. Volume in m^3/m of shoreline; weight in kg/m of shoreline.

	Volume			Weight
Site	Mean	Standard Deviation	Number of lines	Mean
1	84	17	6	87000
2	33	11	12	34000
3	25	22	15	26000
4	22	14	13	23000
11	53	29	15	55000
22	19	19	15	20000
26	34	22	8	35000
38	40	22	9	42000
39	75	25	6	78000
40	47	20	8	49000
46	21	17	15	22000
51	17	11	15	17000
54	34	20	9	35000
56	15	10	7	16000
57	23	11	8	24000
62	32	25	16	34000
63	18	20	16	19000

Table 2. Erosion index (K_e) at selected shoreline sites. Units: tonne-m/m of shoreline/y.

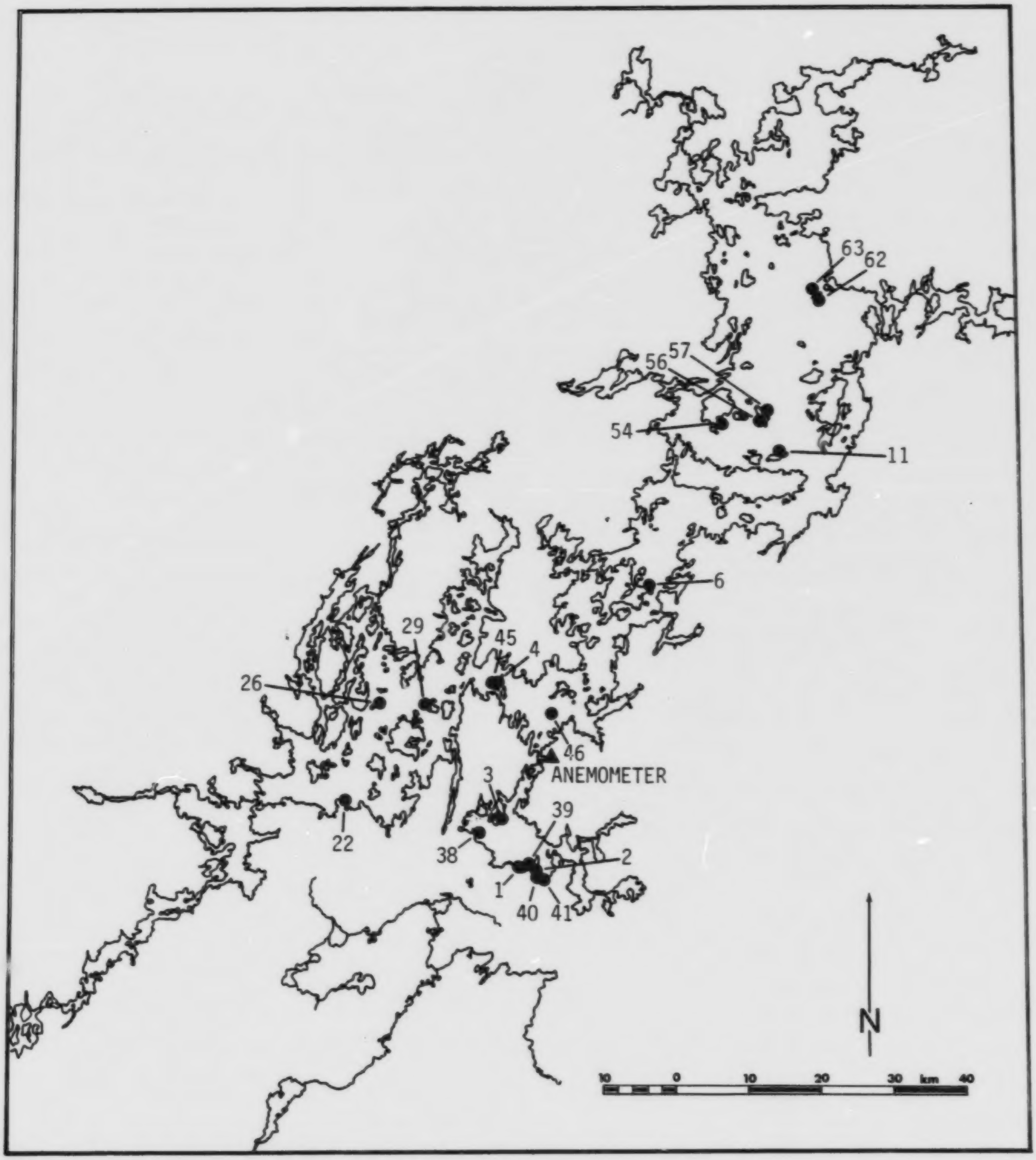
	Erosion Index	
Site	Eroding clay banks	Coarse beach or bedrock at water's edge
1	0.00016	
2	0.00083	
3	0.00019	
4	0.00027	
11		0.00004
22	0.00007	
26	0.00058	
38	0.00054	
39	0.00013	
40	0.00108	
46		0.00006
51	0.00006	
54	0.00023	
56		0.00006
57		0.00003
62		0.00006
63		0.00004
Mean	0.00038	0.00005
Standard Deviation	0.00032	0.00001
Number of sites	11	6

Table 3. Mineral sediment deposited within 100 m of the water's edge at selected shoreline sites. Volume in m^3/m of shoreline; weight in kg/m of shoreline.

Site	Volume			Weight
	Mean	Standard Deviation	Number of lines	Mean
1	28.3	10.2	3	27500
2	16.7	2.1	3	16200
3	16.3	8.1	3	15900
4	14.5	6.5	2	14100
11	1.8	1.1	4	1700
22	16.0	6.5	3	15600
26	19.5	11.3	4	19000
38	31.7	18.2	3	30800
39	28.0	0.0	1	27200
40	34.5	0.5	2	33500
51	5.0	2.5	4	4900
56	4.3	0.5	3	4200
57	0.1	0.1	3	100
62	7.7	3.3	3	7500
63	2.0	0.8	3	1900

Table 4. Ratio of weight of nearshore deposits to weight of eroded bank material at selected sites.

Site	Deposition: Erosion (weight ratio)	
	Eroding clay banks	Coarse beach or bedrock at water's edge
1	0.32	
2	0.47	
3	0.60	
4	0.61	
11		0.03
22	0.78	
26	0.54	
38	0.74	
39	0.35	
40	0.68	
51	0.28	
56		0.27
57		0.00
62		0.22
63		0.10
Mean	0.54	0.13
Standard Deviation	0.17	0.10
Number of sites	10	5


63
62
57
56
54
11
6
29
45
4
26
46
ANEMOMETER
3
39
22
38
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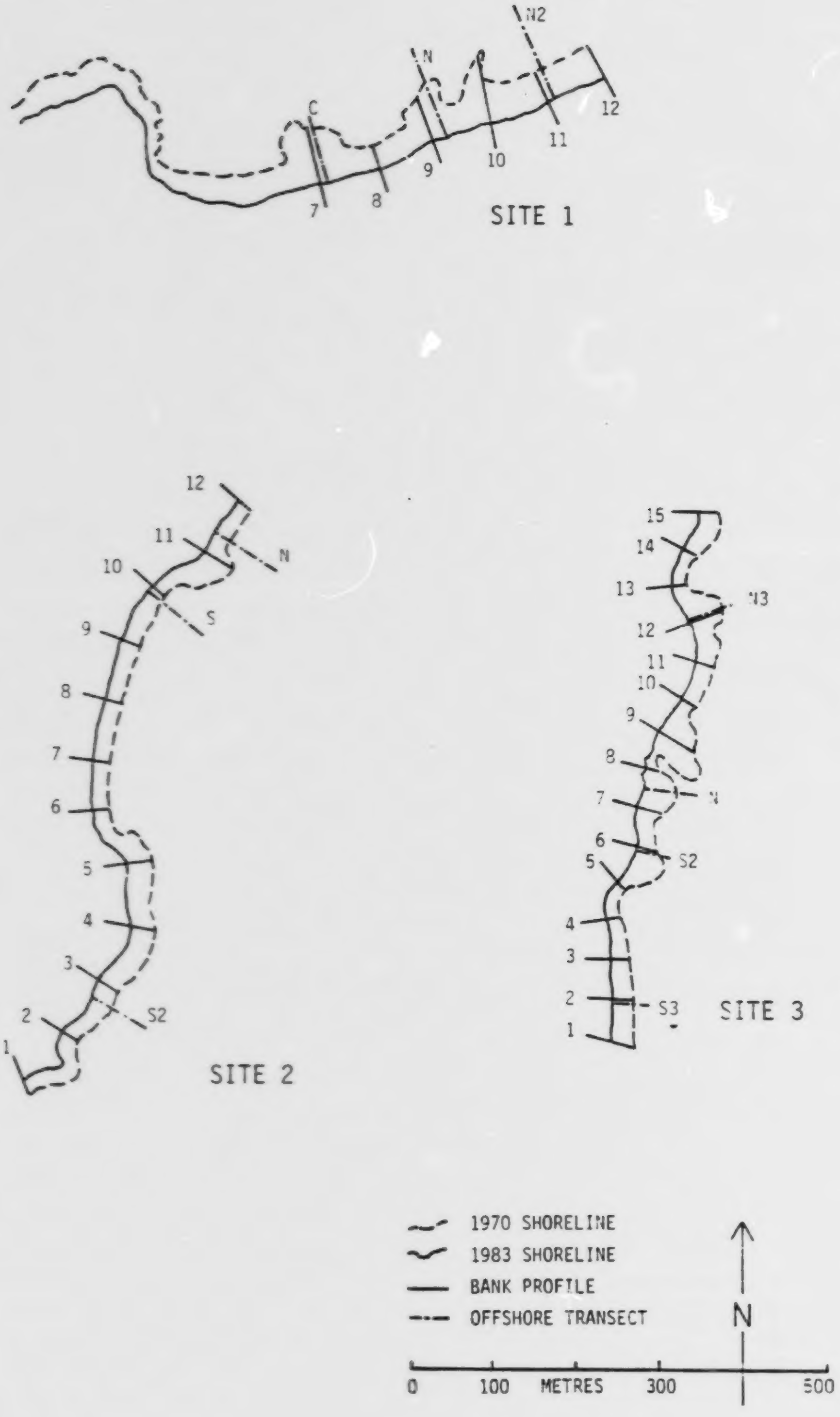
1
40
41
N
10
0
10
20
30
km
40

Figure 1: Shoreline evolution study location map.


N2
N
C
12
11
10
9
8
7
SITE 1
12
11
N
10
S
9
8
7
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5
4
3
2
S2
1
SITE 2
15
14
13
N3
12
11
10
9
8
7
N
6
S2
5
4
3
2
S3
1
SITE 3
1970 SHORELINE
1983 SHORELINE
BANK PROFILE
OFFSHORE TRANSECT
N
0
100
METRES
300
500

Figure 2: Shoreline evolution study site maps.

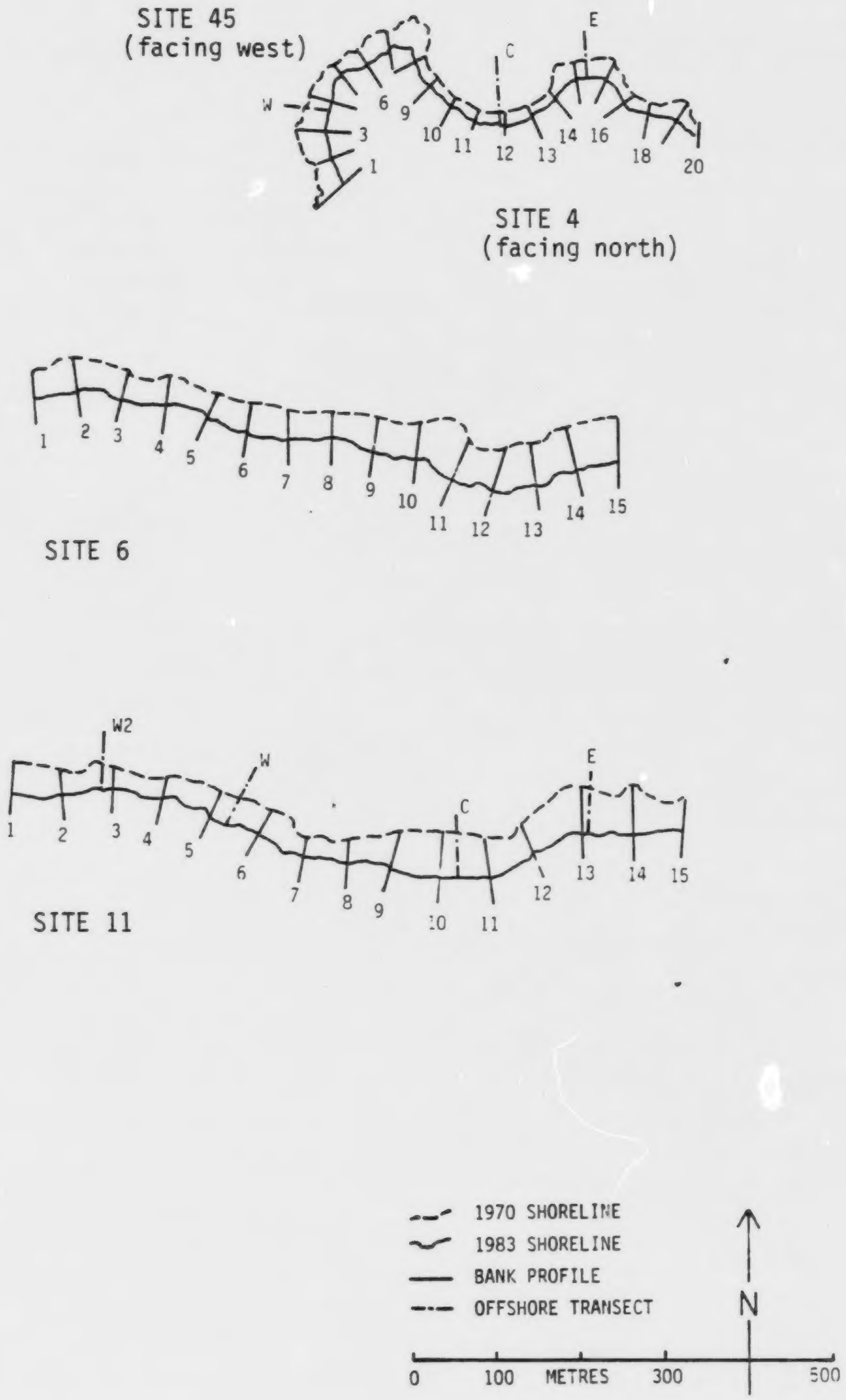
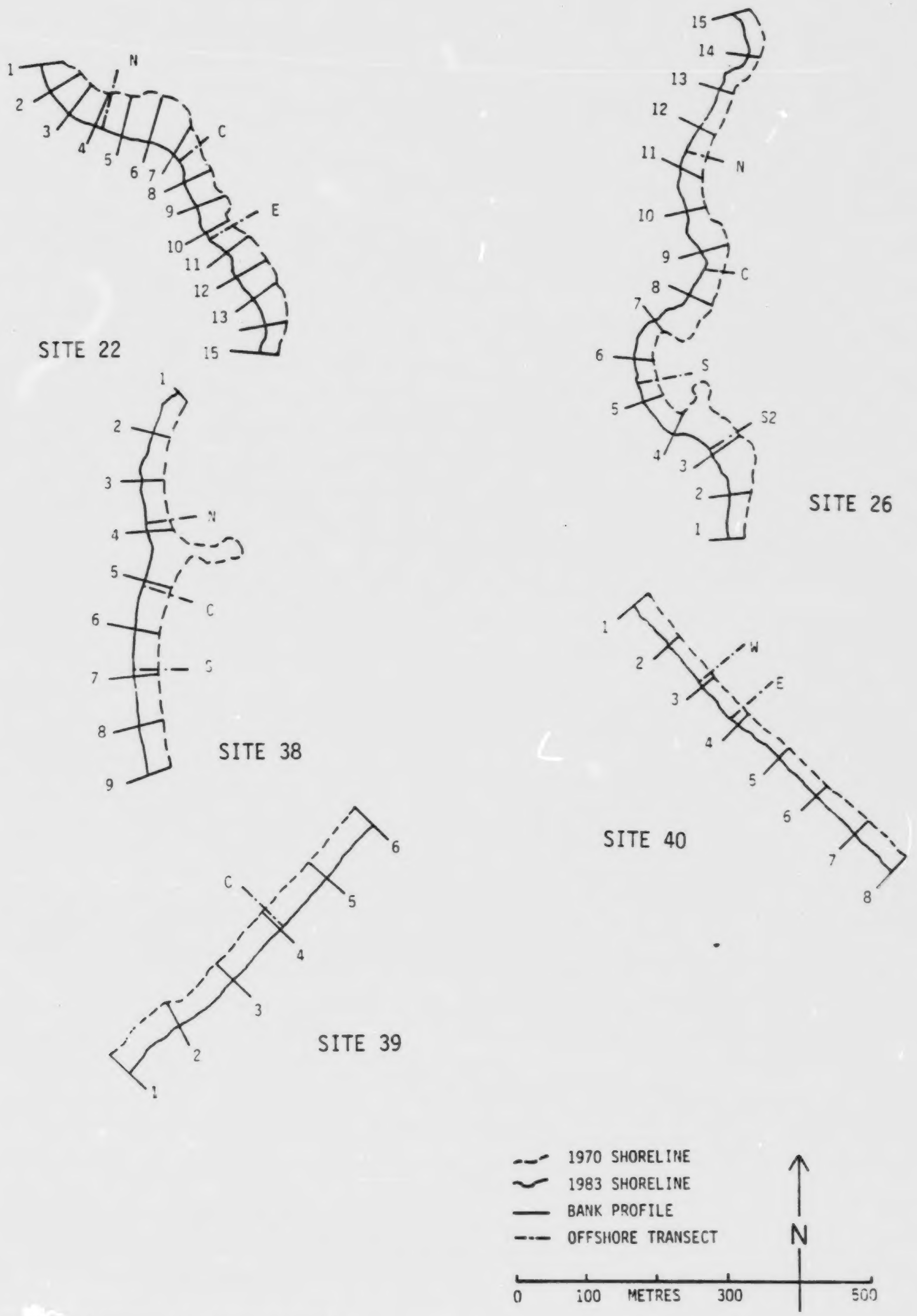

SITE 45
(facing west)
W
C
E
1
3
6
9
10
11
12
13
14
16
18
20
SITE 4
(facing north)
1
2
3
4
5
6
7
8
9
10
11
12
13
14
15
SITE 6
W2
W
C
E
1
2
3
4
5
6
7
8
9
10
11
12
13
14
15
SITE 11
1970 SHORELINE
1983 SHORELINE
BANK PROFILE
OFFSHORE TRANSECT
N
0
100
METRES
300
500

Figure 2: (continued)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SITE 38
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SITE 39
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1983 SHORELINE
BANK PROFILE
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METRES
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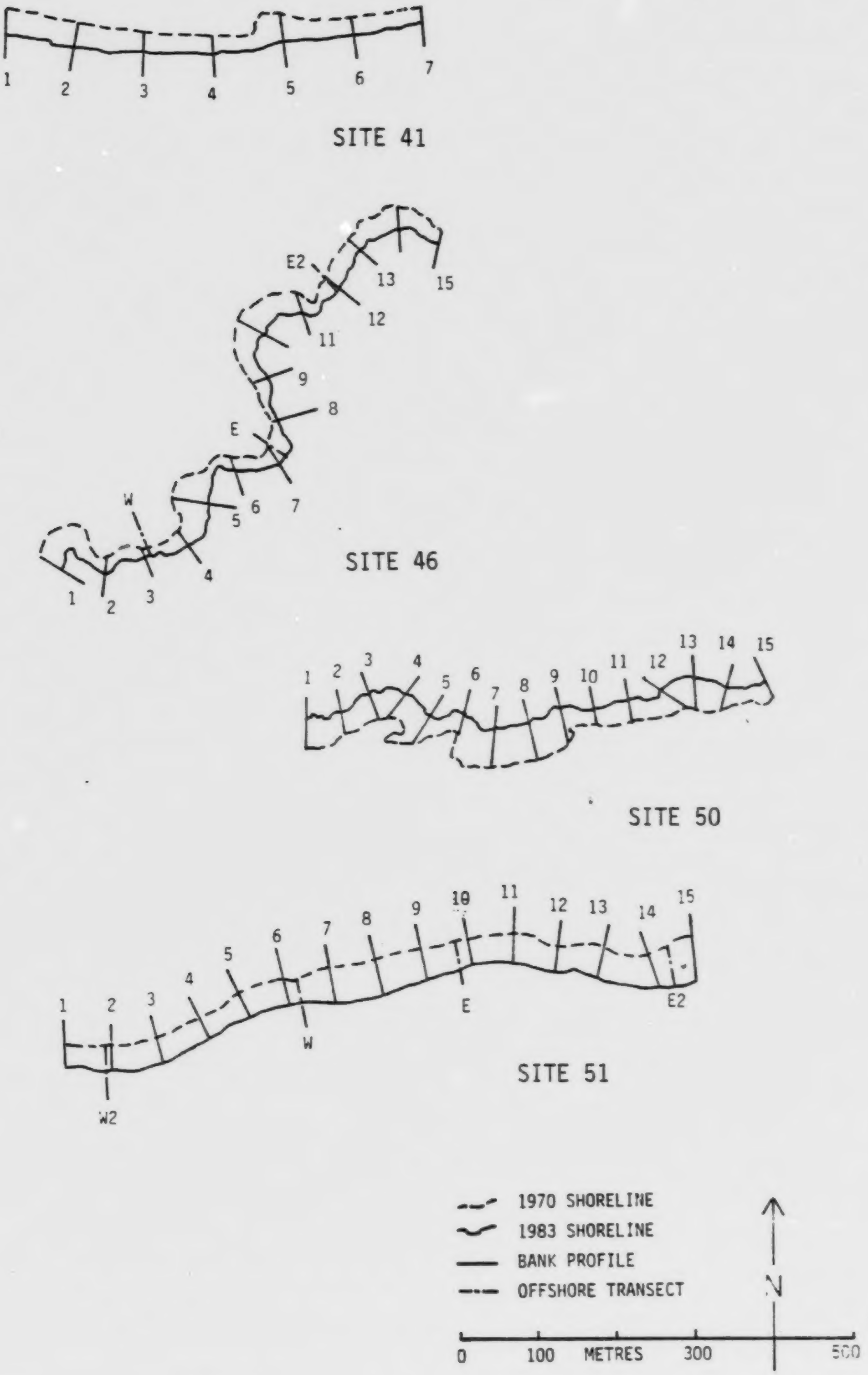
Figure 2: (continued)



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SITE 46
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15
SITE 50
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7
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W
E
E2
W2
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BANK PROFILE
OFFSHORE TRANSECT
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METRES
300
500

Figure 2: (continued)

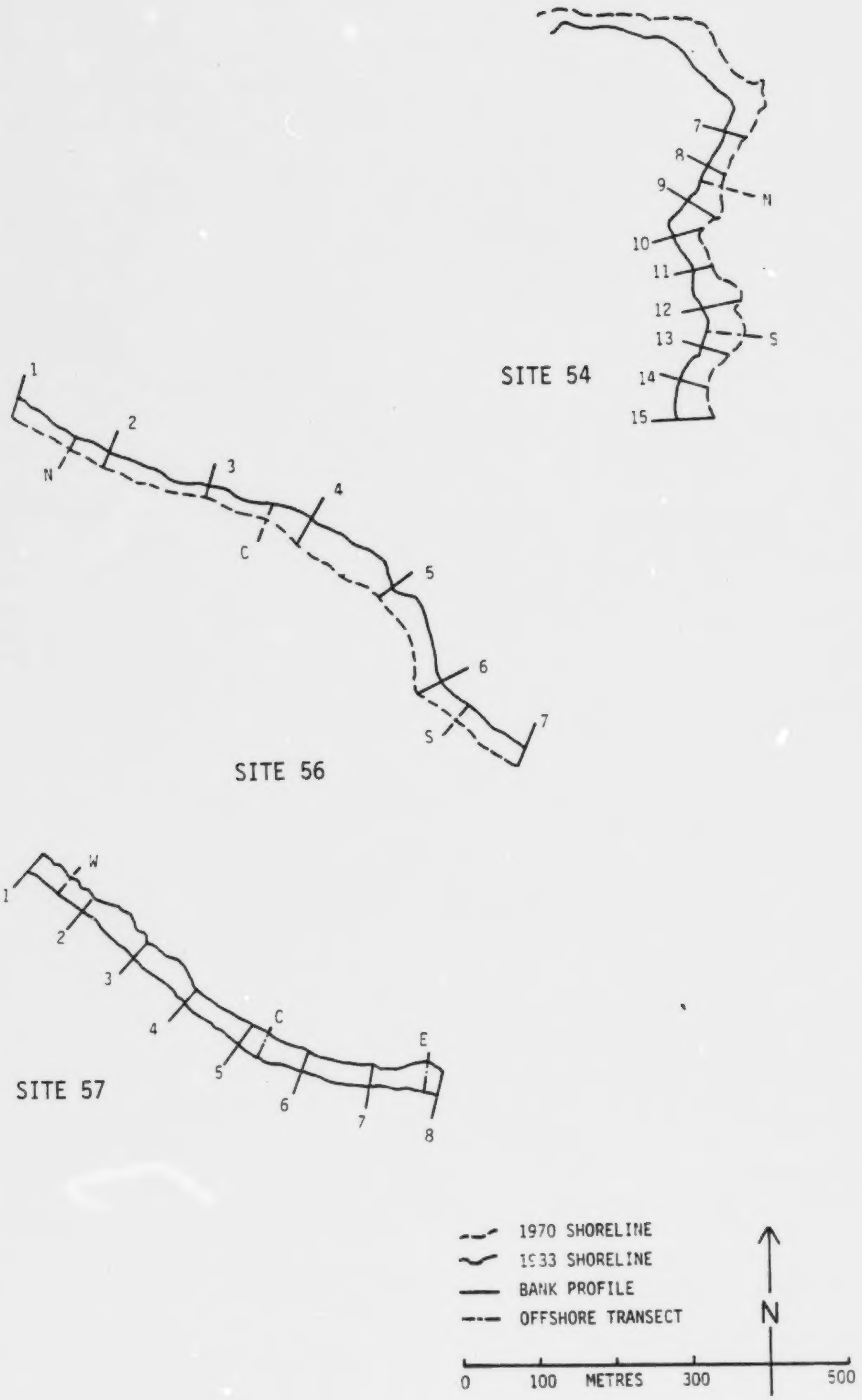

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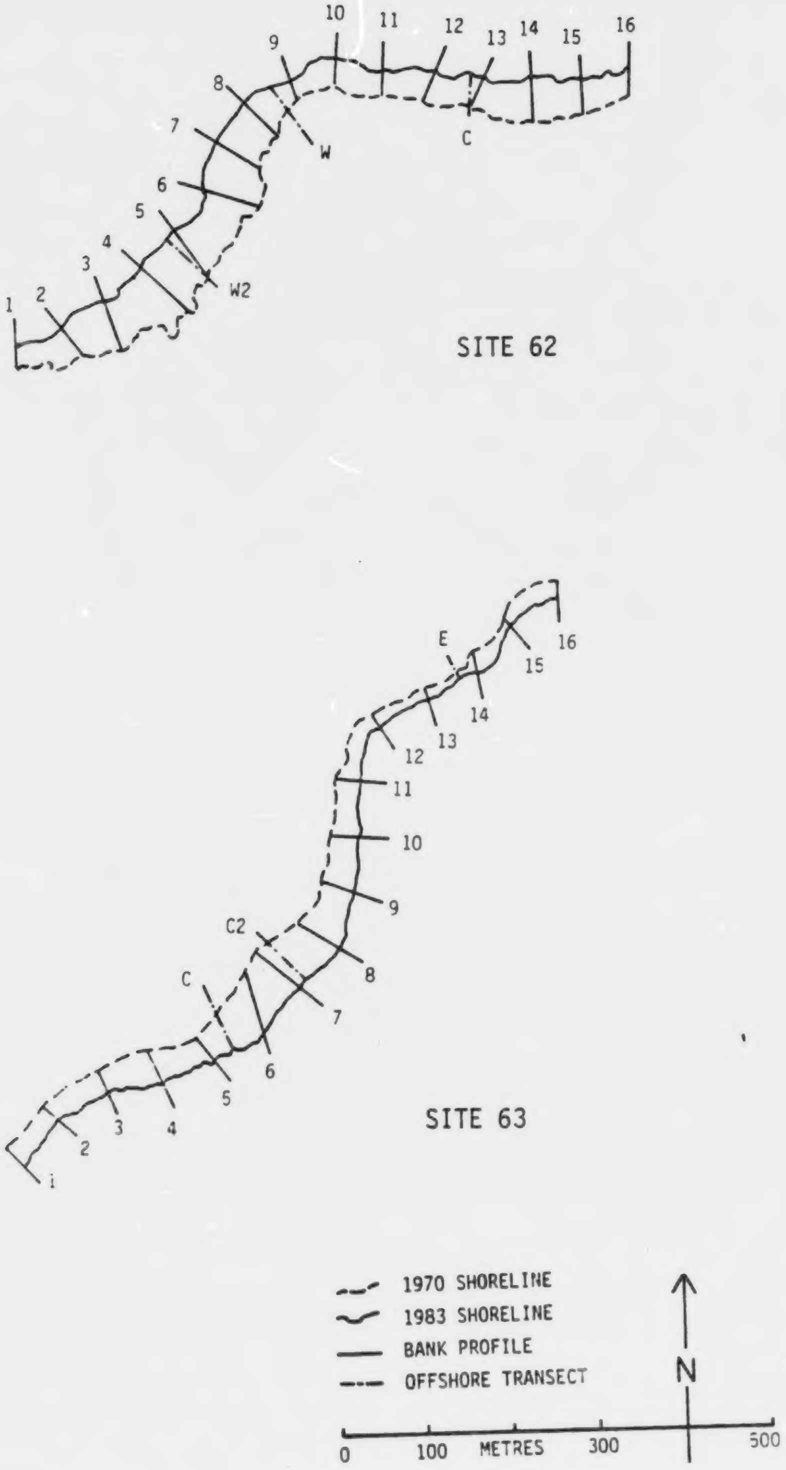
Figure 2: (continued)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C2
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C
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6
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2
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METRES
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500

Figure 2: (continued)

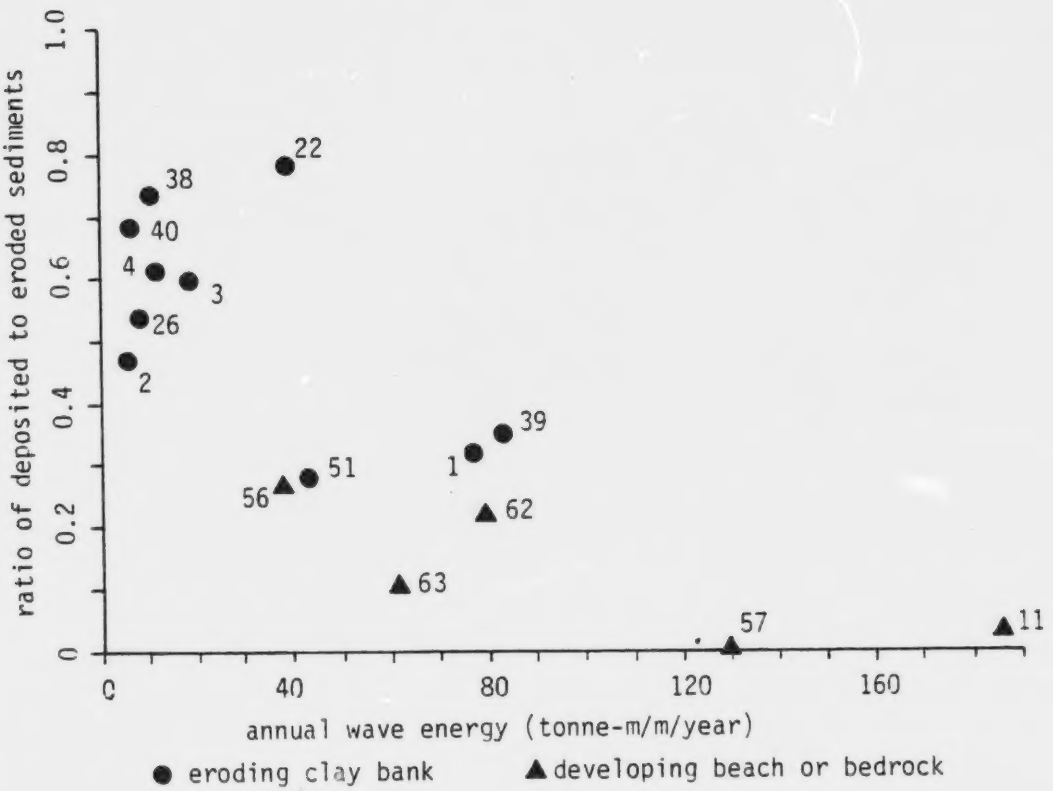

Figure 3: Relationship between ratio of deposited to eroded sediments ($R_{d:e}$) with annual wave energy.

Appendix 3: Measurements of mercury methylation balance in relation to concentrations of total mercury in northern Manitoba reservoirs and their use in predicting the duration of fish mercury problems in new reservoirs

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December, 1986

SPECIAL NOTE

This report was prepared for the Canada-Manitoba Agreement on the Study and Monitoring of Mercury in the Churchill River Diversion. Any conclusions, recommendations, or opinions expressed herein are those of the author and are not necessarily the same as those of the Steering Committee for this Agreement.

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Introduction

The impoundment of Southern Indian Lake (SIL) and diversion of the Churchill River into the Rat and Burntwood River valleys in northern Manitoba (Newbury et al. 1984) resulted in elevated fish mercury levels in the flooded lakes (Bodaly and Hecky 1979; Bodaly et al. 1984) and created concern about commercial marketing and domestic consumption. A prediction of the time course of the fish mercury problem in lakes along the diversion route is clearly desirable, but has not been possible to date. Mercury levels in predatory fish species in SIL have not declined in the 8 years since impoundment. Concentrations of mercury in whitefish have declined somewhat but have not returned to pre-impoundment levels and the direction of any continuing trend is unclear (Bodaly unpubl. data). With the relatively recent introduction in Canada of routine testing for mercury in fish (1970), time course data for older reservoir developments are not available. The objective of this study is to establish a time course for the fish mercury problem in lakes flooded by the Churchill River diversion by examining the balance between mercury methylation and demethylation in reservoirs of different ages.

The majority of mercury in fish muscle is in the organic methyl mercury form yet very little of the mercury in the sediments and water of lakes and rivers and in the soils and vegetation flooded during reservoir formation is methylated (Westoo 1966; Fimreite et al. 1971). This transformation is predominantly bacterially mediated (Beijer and Jernelov 1979; Bisogni 1979). Consequently, there must be a net increase in the rate of bacterial methylation for fish mercury levels to increase and Ramlal et al. (1986 a and b) have found elevated methylation rates in nearshore sediments of lakes flooded by the Churchill River diversion. Bacterial methylation can be stimulated by a number of factors but those which are of primary importance are increased sediment nutrient (organic) content (Jensen and Jernelov 1969; Fagerstrom and Jernelov 1972; Wright and Hamilton 1982; Nagase et al. 1982) and elevated inorganic mercury concentrations (Wright and Hamilton 1982; Nagase et al. 1982).

Sediment organic content may be increased by the flooding of terrestrial vegetation, peat, and humus. These additions can affect methylation directly by stimulating bacterial growth (Wright and Hamilton 1982) and indirectly by altering pH (Fagerstrom and Jernelov 1972; Langley 1973; Nagase et al. 1982) or by reducing oxygen concentrations (Bisogni and Lawrence 1975; Olson and Cooper 1976; Ramial 1983) as a result of the increased oxygen demand. Mercury concentrations may increase in reservoirs if the flooded vegetation and soil have higher mercury levels than in the pre-existing lake or river sediments. The mercury content of organic soils and vegetation flooded by the Churchill River diversion is about three times that of lake sediment (Bodaly et al. 1984).

Increased mercury concentrations have generally been proposed as the cause of elevated fish mercury levels in new reservoirs (Abernathy and Cumbie 1977; Meister et al. 1979), although this hypothesis has never been tested. More recently, Bodaly et al. (1984) have suggested that it is the enhancement of bacterial methylation by the addition of organic substrate from flooded shorelines which effects the observed increases in fish mercury. In either case, as the material introduced to the reservoir during flooding is processed over time, its ability to stimulate methylation should decrease and in turn result in lower fish mercury levels. Therefore, by measuring the net rate of bacterial methylation in reservoirs of different age, it should be possible to estimate the time course of the fish mercury problem in lakes along the Churchill River diversion.

Aquatic methyl mercury concentrations are regulated by the concurrent processes of its production and degradation. Methyl mercury is produced by the methylation of the mercuric ion (Hg^{2+}) (DeSimone et al. 1973) and can be enzymatically degraded to Hg^{o} and methane (Tezuka and Tonomura 1976, 1978; Fox and Walsh 1982). The methods used in this study for mercury methylation and demethylation measurements measured the specific rates of each process (Furutani and Rudd 1980; Ramlal et al. 1986b). The ratio of specific methylation to demethylation (M/D), the methylation balance, was calculated from these rates and is used as an indicator of the net rate of methyl mercury production (Ramlal et al. 1986b).

This paper describes a survey of several reservoirs in northern Manitoba conducted during the summer of 1984 to establish: 1) the location at which specific methylating activity is highest in reservoirs; 2) which of the processes regulating methyl mercury concentrations is most affected by flooding; 3) the relationship between the methylation balance and concentrations of total mercury in the water column and sediments; and 4) a relationship between reservoir age and enhanced mercury methylation.

Measurements of mercury concentrations in water samples have proved inadequate for assessment of the effects of the Churchill River diversion on water column mercury levels, with concentrations generally near or below the limit of detection (5 ng L^{-1}) (Bodaly et al. 1984). This has been noted in other new reservoirs where fish mercury levels were elevated (Cox et al. 1979; Potter et al. 1975; Abernathy and Cumbie 1977) and is expected because the geochemistry of inorganic mercury favours association with sediments (Cranston and Buckley 1972; Hannan and Thompson 1977), and the biogeochemistry of methyl mercury strongly favours association with biota due to its aqueous and lipid solubility and affinity for sulfhydryl groups (Carty and Malone 1979). As an alternative to the measurement of water mercury levels, mercury concentrations in plankton samples were used in this study to provide a relative measure of the effects of reservoir formation and age on the concentration of mercury in the water column. This method has frequently been used to identify aquatic habitats receiving elevated mercury loading (Sarkka et al. 1978 a and b; Skei et al. 1976; Windom et al. 1973; Hirota et al. 1979). In enclosure experiments, Rudd and Turner (1983a) have also shown that ^{203}Hg levels in plankton were proportional to concentrations in the water.

Study Area

Five reservoirs (Southern Indian Lake, Notigi Reservoir, Stephens Lake, Sipiwesk Lake, and Reindeer Lake) and two control lakes (Granville and East Mynarski) (Table 1; Fig. 1) were included in the survey. The reservoirs range in age from 8 to 42 years and flooded terrain accounts for 0.5 to 100% of the reservoir area. Granville Lake is part of the Churchill River system and East Mynarski Lake is proximal to Notigi Reservoir but neither has been affected by the diversion. All of the lakes/reservoirs lie in the Precambrian Shield, are north of the southern limit of permafrost, and all are located in areas of glaciolacustrine sedimentation (Fig. 2). All of the basins lie in the wide band of boreal forest (taiga) which crosses mid-latitude Canada and are dominated by black spruce (Picea mariana) so the same types of vegetation were flooded in all of the reservoir developments. More detailed descriptions of the local geography, geology, and vegetation may be found in Newbury et al. (1984), Davies et al. (1962), Frohlinger (1972), Ritchie (1962), and Water Resources Branch (1974).

Methods

Location and frequency of sampling

A total of 9 sites, two in each of SIL and Notigi Reservoir and one in each of Stephens, Reindeer, Sipiwesk, East Mynarski, and Granville Lakes were surveyed 5 times between June 25 and September 28, 1984 (Table 2; Fig. 1). Generally, sediment samples for measurement of methylation and demethylation rates were collected from two stations in the reference lakes and from three stations at each of the reservoir sites. Three or four additional stations were sampled at each site on the survey in early August to provide a measure of local spatial variability. Stations were classified as inshore or offshore and as flooded or unflooded. Flooded stations were located over inundated terrestrial material in the reservoirs as determined from depth measurements, the estimated depth of flooding (Brown 1974; Water Resources Branch 1974), water level records (Water Survey of Canada provisional data; Manitoba Hydro unpubl. data), and examination of bottom material collected with an Ekman dredge. In the reservoirs, unflooded stations were located over fine-grained sediments within the pre-flooding lake basin. Although Sipiwesk Lake is a reservoir, no flooded sites could be accessed because of low water levels. Inshore stations were generally within 5 metres of the lake shore except at SIL Region 5 where samples were collected about 20 metres from shore because flooded trees prevented closer access by the float plane. Offshore stations were generally 300-400 metres directly offshore from the inshore stations at each site.

Samples collected

On the five surveys, mercury methylation and methylmercury demethylation rates were measured on samples of surficial lake bottom sediment collected from each station. The samples (2L volume) were collected using a 12 volt submersible bilge pump (Attwood model 4103, 360 gallons per hour) attached to a 15 m length of vinyl-lined plastic garden hose.

Plankton samples were collected from each site on all surveys for analysis of total mercury concentrations. Four Wisconsin-style nets (25 cm mouth diameter, 1 m in length), constructed with 53, 150, 300, and 700 um mesh-size Nitex, respectively, were towed horizontally for 3-7 minutes (depending on plankton abundance) about 20 m behind the plane/boat between the inshore and offshore stations. The plankton samples collected in each net were bottled separately and kept on ice for return to the field lab. Later the same day, the volume of each sample was reduced by seiving through 50 um mesh-size Nitex and the sample was transferred to a 20 ml plastic scintillation vial or an 8 oz. Whirlpak bag and immediately frozen for later analysis.

Profiles of temperature, conductivity, pH, and dissolved oxygen concentration were recorded at the deepest station at each site on all surveys with an Hydrolab model 4041 Water Quality Monitor. Water transparency was measured at each station with a 20 cm diameter black and white Secchi disc.

Laboratory Analyses

Subsamples of the collected sediment were taken for determination of pH and dry weights and for analysis of particulate carbon (as total carbon, organic carbon, and carbonate carbon), nitrogen (as total nitrogen) (Stainton et al. 1977), and total mercury (Armstrong and Uthe 1971) concentrations. The remaining sediment was used for methylation and demethylation measurements.

Isotope addition and sample incubation

In the laboratory, the sediment was mixed and 100 mL measured into acid washed 125 mL glass reagent bottles. Three replicate subsamples were taken of each sample, one blank and a pair of replicates, for each methylation and demethylation determination. The microbial activity of the sediment blanks was stopped with 1 mL of 4N HCl prior to isotope addition. Methylation samples were injected with ^{203}Hg as $HgCl_2$ at a dose of 74kBq (2ug Hg)$^{-1}$ (g dry sediment)$^{-1}$. Demethylation samples were injected with ^{14}C as CH_3HgI at a dose of 14.8kBq (9.17 ug Hg)$^{-1}$ (g dry sediment)$^{-1}$. The samples were shaken for one minute after isotope addition then incubated in the dark at in situ temperature for 12 hours. Sediments collected from sites where the overlying water was anoxic were incubated with nitrogen in the head space of the incubation vessel to maintain anaerobic conditions. All other samples were incubated with room air in the head space.

Specific rate of mercury methylation

The specific rate of mercury methylation was determined using the method of Furutani and Rudd (1980). After the samples were incubated and the microbial activity halted with 4N HCl, the alkylated ^{203}Hg was extracted from the sediment using an organic solvent extraction technique (Furutani and Rudd 1980).

Specific rate of methyl mercury demethylation

The specific rate of demethylation was measured using the method of Ramlal et al. (1986b). At the end of the incubation period, the volatile ^{14}C produced by demethylation was stripped from the acidified samples, oxidized to $^{14}CO_2$, and trapped in a carbon dioxide trap consisting of the fluor PCS (Amersham), methanol, and CO_2 mMet, a carbon dioxide trapping agent (Amersham) (Ramlal et al. 1986b).

Calculations

The rate of methylation in sediment was first calculated in ng Hg methylated g^{-1} h^{-1}. This value was then used to calculate the specific rate of methylation, i.e. the percent of the added mercury methylated g^{-1} h^{-1}. The specific rate of demethylation was calculated in a similar manner (Ramlal et al. 1986b). The ratio of specific methylation to specific demethylation (M/D) was calculated from the rates for the two processes:

$$M/D = \frac{\%\ \text{added Hg methylated}\ g^{-1}\ h^{-1}}{\%\ \text{added Hg demethylated}\ g^{-1}\ h^{-1}}$$

Plankton mercury concentration

The plankton samples were dried to constant weight at 60° C, the dry weight of the sample calculated, and the total mercury concentration determined using the method of Armstrong and Uthe (1971). This method proved inaccurate for sample weights below 20 mg dry wt (R. Hunt, Department of Fisheries and Oceans, Freshwater Institute, Winnipeg, Manitoba, pers. comm.). The weights of samples collected in the 300 and 700 um mesh-size nets were generally below this limit so the discussion of plankton mercury concentrations was restricted to the 53 and 150 um size-fractions. Methyl mercury concentrations were also determined, using the method of Uthe et al. (1972), on at least one sample collected from each site on the fourth survey.

Results

A broad range of water transparencies and near-bottom conductivities were represented in the lakes/reservoirs included in the survey, with the control lakes occupying an intermediate position (Table 3). Near-bottom pH was in the circumneutral range at all sites (Table 3). On average, the near-bottom temperature was coolest at the Issett and SIL Region 5 sites and was warmest at the Sipiwesk L. and Granville L. sites (Table 3). Region 5 was also the coolest at the start of sampling while Sipiwesk L. was the warmest at this time, with an 11° C temperature differential between the sites (Fig. 4). In general, the intersite range of temperatures decreased from spring to autumn (Fig. 4). Water temperature peaked in early August at all sites (Fig. 4). Thermal stratification was evident only at the Issett L. site and persisted from the start of sampling on June 27 until early August, breaking down following the onset of autumn cooling. The bottom waters were well-oxygenated at all sites except those in Issett L. (Table 3), where values less than 2.0 ppm were measured on all but the final survey.

The rate of biological mercury methylation followed a distinct seasonal pattern at both the inshore and offshore stations in all of the lakes/reservoirs studied (Fig. 5 and 6). The highest rates occurred in August, generally during the first half of the month, with low to undetectable methylation rates at the beginning and end of the sampling period. There was no consistent seasonal trend in the rate of demethylation (Fig. 7 and 8) among the lakes/reservoirs so the M/D ratio followed a seasonal pattern which paralleled that of methylation (Fig. 9 and 10).

On average, specific methylation rates at inshore stations were markedly higher (range: 1.7 to 8.3 times higher) in reservoirs where the samples were collected from flooded terrain than in the control lakes (Table 4). In Sipiwesk L., where the inshore stations were located within the preflooding lake basin, the seasonal mean methylation rate was below that in either of the control lakes (Table 4). There was no relationship between methylation rate at the inshore sites and reservoir age. The seasonal mean rate at the Reindeer L. site was within the range of values measured at the SIL sites (Table 4). In general the methylation rates at flooded sites were strikingly similar, the only exception being the four-fold higher rates measured at the Issett L. site (Table 4).

In contrast to the elevated methylation rates at inshore stations over flooded material, seasonal mean specific demethylation rates were either the same as (SIL Region 5) or were lower in the reservoirs than in the control lakes (Table 4). As a result, the M/D ratio was elevated to a greater degree than was methylation alone at all but the Region 5 site (Table 4). The mean demethylation rate at the inshore stations in Sipiwesk L. was comparable to the control rates (Table 4).

In extensively flooded reservoirs where the offshore stations were located over flooded material (W. Notigi L., Issett L., and Stephens L.), the specific methylation rates were as high or higher than at the corresponding inshore stations (Table 4). At the Methyl Bay and Region 5 sites, where the offshore stations were located outside the flooded zone, the methylation rates were considerably lower than at the inshore stations with the difference between the inshore and offshore rates greater in Methyl Bay than in Region 5 (Table 4). Although the offshore stations in Reindeer L. were also located outside the flooded zone, the methylation rate was similar to that at the inshore stations (Table 4). In Granville L., the seasonal mean methylation rate at the offshore station was 8 times lower than at the inshore station and was lower than at any site except Methyl Bay (Table 4). In contrast, the methylation rate at the offshore station in E. Mynarski L. was 4 times higher than at the inshore station and was higher than all sites except Stephens L. and Issett L. (Table 4).

Rates of specific biological demethylation at the offshore stations were generally comparable to or lower than the demethylation rates at the inshore stations (Table 4). Again, the only exception to this pattern was the E. Mynarski site, where the demethylation rate at the offshore station was about twice the inshore rate and was 3 to 5 times the offshore rate at any of the other sites (Table 4).

The high specific rates of methylation and demethylation at the offshore station in E. Mynarski L. were largely compensatory and the methylation balance (M/D), which is an indication of the net rate of mercury methylation, was comparable to that in Granville L.. As a result it is necessary to use the M/D ratio to compare methylating activity among the offshore stations while consideration of the specific methylation rates alone was sufficient at the inshore stations. Nevertheless, a similar picture of the effects of flooding on the methylation balance emerges. The M/D ratios at offshore stations located within the flooded zones of the reservoirs and at the offshore station in Reindeer L. ranged from 3 to 11 times higher than in E. Mynarski L. which is as great an elevation as at the flooded inshore sites (Table 4).

Considerable spatial variation in specific methylation rates was found on the early August survey and the magnitude of this variation differed substantially among the sites, with standard errors ranging from 31 to 79% of the mean at the inshore stations (Table 5). Specific demethylation rates varied much less at a given site and the degree of spatial variation was much more consistent from site to site (Table 5).

Given the strong dependence of the seasonal mean specific methylation values discussed above on the measurements made on the third survey (Fig. 5 and 6) and the known sensitivity of the mean to extreme values (Huntsberger and Billingsly 1977), the reliability of the mean as a measure of central tendency of methylation rates was evaluated. Because of its insensitivity to outlying values, the trimean is considered to be a better measure of location in situations of high variance (Huntsberger and Billingsly 1977). The trimean is calculated using the values of the lower and upper quartiles and the median of the frequency distribution of the measured variable. With no more than four observations from a particular site-location, a frequency distribution could not be generated for each site. Instead, the measurements from all similar site-locations were pooled to produce the required frequency curves. The two groups of sites selected were the inshore stations in the control lakes and the flooded inshore stations in the reservoirs. The Issett L. site, with its notably higher methylation rates, was excluded from this second group. The greater frequency of higher methylation rates and the higher maximum methylation rates at the reservoir sites were evident (Figure 11). The trimean specific methylation rate for the reservoir sites was 4.6 times the control rate while the mean rate for the reservoir sites was only 2.6 times the control rate (Table 6). This indicates that the higher mean methylation rates at the inshore reservoir stations reflect the general trend of the data and are not the product of a few extreme values. Furthermore, the difference between arithmetic means is a conservative estimator of the effect of flooding on mercury methylation.

Among the reservoirs, total mercury concentrations in sediments were markedly higher than in the control lakes at just two sites, Methyl Bay and Issett L. (Table 7). At Methyl Bay, this elevation was more apparent on the first survey than on the fourth while the higher levels at the Issett L. site were evident only on the first survey (Table 7). There was no significant correlation between the seasonal mean sediment total mercury concentration and the seasonal mean specific rate of mercury methylation ($r=-0.11$, $p>0.10$).

Total mercury concentrations in plankton followed a distinct and identical seasonal pattern of fluctuation in all of the lakes/reservoirs examined. Levels were uniformly low through the first half of August with a substantial pulse occurring at all sites in the second half of the month (Fig. 12). At all sites, mercury concentrations declined between the fourth and fifth surveys and, except in E. Mynarski L., Reindeer L., and SIL Region 5, had returned to those measured previous to the pulse by late September (Fig. 12). Although this pattern is similar to that of mercury methylation, no methyl mercury was detected in plankton collected from any of the sites on the fourth survey.

Reservoir formation did not cause an elevation of total mercury concentrations in the water column, as indicated by mercury levels in the plankton. On average, plankton mercury levels in the control lakes were as high or higher than at any of the reservoir sites except Reindeer L. (Fig. 13). As a result, there was no significant relationship between mercury methylation rate and water column total mercury as estimated by the plankton total mercury concentration ($r=0.22$, $p>0.10$). There also was no significant difference in the total mercury content of plankton collected in the 53 and 150 um mesh-size nets (t-test, $p>0.10$).

Discussion

The present study has identified several northern Manitoba reservoirs, and specifically the flooded zones of these reservoirs, as sites supporting elevated rates of biological net mercury methylation as indicated by elevated M/D ratios. Our results are consistent with those of Ramlal et al. (1986a and b). and indicate the phenomenon is not restricted to lakes along the Churchill River diversion. However, while Ramlal et al. (1986a and b) found higher rates of both methylation and demethylation at flooded sites, with the elevated M/D ratios the result of a greater increase in methylation, the elevated M/D ratios measured in this study were almost solely the result of higher rates of methylation in the reservoirs. Seasonal mean rates of biological degradation of methyl mercury were never higher in the reservoirs than in the control lakes (Table 4). This discrepancy may be a combined result of the considerable and asynchronous temporal variation of demethylating activity observed in all of the lakes/reservoirs in this study and that the previous studies relied upon a few points in time measurements of methylation and demethylation.

The occurrence of elevated net methylation rates provides an explanation for the observed increases in fish mercury levels following the flooding of Southern Indian Lake and the formation of Notigi Reservoir (Bodaly and Hecky 1979; Bodaly et al. 1984; Table 8) and Stephens Lake (Table 8). In the absence of preimpoundment data for Reindeer Lake it is not known if mercury levels in fish were elevated following impoundment. The present mercury level in northern pike from Reindeer Lake (Table 8) is close to the median for lakes in northern Manitoba (0.43 ug/g, Southern Operations Directorate, Department of Fisheries and Oceans, Winnipeg, MB.) which indicates that the levels are not unusually high despite our finding elevated methylation rates at the site investigated. In flooded lakes along the Churchill River diversion, fish mercury levels were found to be proportional to the flooded terrestrial area as a percentage of the preimpoundment lake area (Bodaly et al. 1984). Considering that the area of Reindeer L. only increased 0.5% due to flooding (Water Resources Branch 1974), mercury levels in fish from the lake may never have been much higher than at present despite elevated rates in and near the flooded zone.

The high M/D ratio at the offshore station in Reindeer L. should not be taken as an indication that rates of methyl mercury production are elevated all over the lake. The sampling site was located in a small, shallow (Z_{max} 5m), sheltered, bay to permit access in all weather. The bay also was isolated from the main basin of Reindeer L. by a shallow (2 m depth) sill. In this situation, material eroded from the shoreline could be expected to be deposited within the bay but outside the flooded zone (Appendix 2). As such, the Reindeer L. sampling site probably represents a special case; the small size of the bay means it is influenced to a much greater degree by the flooded shoreline than the rest of the lake and its sheltered location permits accumulation of flooded materials in the offshore zone. Given the small extent of flooding overall, offshore deposition is not likely to be important in most of the lake and this is supported by the mercury levels in fish which are not demonstrably elevated and by the measurements at the more exposed offshore sites in SIL where net methyl mercury production was not elevated relative to the reference lakes (Table 4).

In all but one instance, the seasonal mean rates of methylation at reservoir stations located over flooded terrain were markedly higher than at a corresponding location in either of the unflooded control lakes (Table 4). This exception was the result of the high specific methylation rate measured at the offshore station in E. Mynarski L. and illustrates the importance of the methylation balance and not just the methylation rate in regulating fish mercury levels. Demethylating activity in this part of the lake was compensatorily high so the resulting M/D ratio was near that in Granville Lake and at the unflooded stations in Sipiwesk and Southern Indian Lakes (Table 4). Mercury levels are also low in fish from E. Mynarski L. (Table 8), reflecting the methylation balance rather than the methylation rate.

Consistent with its status as a reservoir, Sipiwesk L. has a history of high fish mercury levels (Table 8). In contrast, the net rate of methyl mercury production in sediments from Sipiwesk L., as indicated by the M/D ratio, was among the lowest measured in any of the lakes/reservoirs surveyed (Table 4). The low methylating activity would appear to be the result of the low water levels which prevented the collection of sediments from flooded areas. However, the samples were collected near the flooded zone and the offshore methylation rates for Reindeer L. site indicate that somewhat higher rates might be expected in Sipiwesk L. sediments (Table 4). Organic matter which appeared to be terrestrial in origin was found in dredge samples taken from Sipiwesk L. but this material was covered by several centimetres of inorganic clay. Sipiwesk L. was the most turbid of the reservoirs included in the survey (Table 3). Jernelov (1970) and Rudd and Turner (1983b) found that the covering of methyl mercury producing sediments with nutrient poor sediments (eg:clay) effectively reduced the rate of methyl mercury production so any stimulatory effect of the terrestrial material on methylation might be negated by this clay covering.

Mercury in the water column

The concentration of mercury in plankton was used as a relative measure of the water mercury concentrations in the lakes and reservoirs in the absence of a method with sufficient sensitivity to measure the water mercury concentration directly. The plankton mercury concentrations measured in this study (Fig. 13) are all within the range of the few measurements which have been made in uncontaminated lakes in Canada (Environment Canada 1979). The levels reported here are also just 20% of the concentrations found in Lake Michigan plankton in 1970 (Copeland 1972), which was not considered to be mercury contaminated at that time. Background mercury levels in marine plankton are similarly low, ranging from 0.20 to 0.35 ppm (dw) (Knauer and Martin 1972; Cocoros et al.

1973; Skei et al. 1976). It is apparent from the similarity of mercury levels measured in the different water bodies in this study, and the comparability of these levels with those in uncontaminated systems, that flooding has not caused an increase in the water mercury concentrations in any of the reservoirs included in the survey nor are concentrations unusually high.

The lack of a difference in mercury concentration between the 53 and 150 um size-fractions is in keeping with the results of Sarkka et al. (1978a), who found no difference in mercury concentration among plankton samples collected with 25, 50, 100, and 350 um mesh-size nets. These findings are also consistent with the results of other studies which suggest there is no food chain amplification of metal concentrations within the plankton. Watras et al. (1985) found that direct uptake from solution rather than uptake from ingested algae was the primary vector for accumulation of nickel by Daphnia magna. Furthermore, D. magna accumulated less nickel per unit of biomass than did the alga Scenedesmus obliquus. Similarly, Hatakeyama and Yasuno (1981) found that Moina sp. accumulated cadmium directly from solution rather than from Cd-loaded Chlorella sp.. Skei et al. (1976) found that mercury levels in phytoplankton-dominated samples from a Norwegian fjord were higher than in zooplankton-dominated samples. Knauer and Martin (1972) and Cocoros et al. (1973) also reported lower concentrations of mercury in marine zooplankton than in phytoplankton from the same area. Although counter to the traditional concept of food-chain amplification of environmental contaminants, it is precisely this absence of bioamplification that is required for mercury concentrations in the mixed plankton collections made in this study to be indicative of concentrations in the water column.

To our knowledge, the distinct seasonal fluctuation of plankton mercury concentrations found in all of the lakes and reservoirs sampled (Fig. 12) has not previously been reported. This pattern is superficially similar to the seasonal fluctuation of methylation rates, but the lack of detectable methyl mercury concentrations in the plankton samples collected in late August indicates methylation was not responsible. Rather, the pattern indicates there was an influx of inorganic mercury to the water column of all systems sometime in August.

The potential causes of this mercury pulse are atmospheric deposition, runoff from the watershed, or a release from lake sediments. Emissions from base metal smelters represent the single largest source (63%) of mercury to the atmosphere in western Canada (Environment Canada 1983) with the smelters in Thompson and Flin Flon representing the most important local sources. However, metal deposition from these sources is limited to a radius of a few tens of kilometeres (Wotton and Hogan 1981) so any emissions are unlikely to affect the survey lakes. Furthermore, with the exception of Sipiwesk L., all of the lakes/reservoirs lie upwind of the prevailing winds from the smelters so any emissions should most affect plankton mercury levels in Sipiwesk L., but this was not the case (Fig. 13). Consequently, an increase in the atmospheric deposition of mercury is not a likely cause of the observed late-summer pulse. Similarly, runoff from the watershed is an unlikely source. There is considerable local variation in the mercury levels in bedrock across the study area (Derksen 1978). If mercury is being leached from the watershed, this variation should be evident in the plankton mercury levels but there is none. In contrast, mercury levels in lake sediments vary little over the study area. Therefore, the mercury most probably was released from the lake sediments.

Hecky et al. (1986) found a similar seasonal cycle of dissolved Si in Southern Indian Lake and Lake Winnipeg with the sediments indicated to be the source of the late summer pulse as well. The cause of this release from the sediments was speculated to be fall cooling which promotes thermal instability in the upper interstitial water column causing circulation within the sediments and a flushing of dissolved constituents (Hecky et al. 1986). Consistent with this hypothesis, the mercury pulse appeared in the plankton at all sites shortly after the onset of cooling (Fig. 3 and 11).

Seasonal variation of methylation and demethylation rates

The rate of mercury methylation followed a striking seasonal pattern in all of the lakes/reservoirs studied despite the often large absolute differences in methylation rate (Fig. 5 and 6). In general, peak specific methylation rates were measured during the period of maximum water temperature (Fig. 3, 5, and 6) but methylation was not a linear function of water temperature at any of the sites. Therefore, the seasonal temperature cycle and the resulting effect on microbial activity (Jernelov 1972) provides, at most, a partial explanation for the seasonal cycle of methylating activity.

Furutani and Rudd (1980) found periodic pulses of methylation in Clay Lake, northwestern Ontario, which were attributed to the cyclic supply of phytoplankton-derived bacterial substrates or to periodic resuspension of sediment. Considering the broad regional variation in the seasonal cycle of primary productivity in Southern Indian Lake (Hecky and Guildford 1984) the former is probably not the cause of the cyclic pattern of methylation reported here but sediment resuspension cannot be ruled out. The pulse of mercury in the water column (Fig. 11), which we suggest was the result of sediment flushing, occurred at about the same time as the peak in methylating activity at all of the sites investigated. With no correlation between the size of the mercury pulse and the peak specific methylation rate, increased mercury availability itself may not be the direct cause of the pulse in methylation.

However, some other unmeasured sediment constituent may have become available to the methylating bacteria during this period of sediment flushing. Further studies are planned to better identify the cause of the methylation cycle.

Clearly, the seasonal pattern of methylation must be considered when making comparisons between lakes. Had samples not been taken in midsummer, there would have been little difference in the methylation rates between the reservoirs and control lakes. Similarly, if June rates in a reservoir were compared with early August rates in a control lake, we would have reached the quite erroneous conclusion that methylation rates are reduced by flooding.

No comparable seasonal fluctuation in demethylating activity was observed (Fig. 7 and 8). The degradation of methylmercury is mediated by the enzyme organomercurial lyase (Robinson and Tuovinen 1984) and the genetic information for production of this enzyme is encoded on transposons (sexually transmitted plasmids; Pan-Hou et al. 1980). The synthesis of the enzyme must be triggered by the presence of a mercury containing inducer substance which may or may not be the substrate and which varies depending on the species of demethylator under consideration (for a review see Robinson and Tuovinen 1984). Therefore, demethylation rates will increase only in the presence of one or more microbial populations which contain the plasmids and provided the appropriate inducer is also present in sufficient concentrations. Consequently, demethylation is not simply dependent on microbial activity. Organisms containing the plasmid were present in all of the lakes/reservoirs considered in the study but flooding did not seem to affect the abundance of any requisite inducer substance nor was there a selection for demethylating bacteria.

Cause of the elevated methylation rates

Past studies have focussed on increased supplies of inorganic mercury as the cause of high fish mercury levels in new reservoirs (Potter et al. 1975; Abernathy and Cumbie 1977; Meister et al. 1979; Cox et al. 1979). These studies either disregarded the fact that incoming mercury has to be methylated to accumulate in fish or implicitly assumed that methylation was limited by mercury availability. While it is well known that the concentration of Hg^{2+} can affect the rate of methylation (Rudd et al. 1983; Wright and Hamilton 1982) it has yet to be demonstrated that any such limitation is alleviated by reservoir formation.

In this study, there was no correlation between methylation rate and the concentration of total mercury in either sediment or water (as indicated by the concentration in plankton). Although terrestrial vegetation and organic soils in the area have higher total mercury concentrations than are commonly found in lake sediments (Bodaly et al. 1984), the introduction of these materials by flooding and shoreline erosion has not produced elevated water total mercury levels in any of the reservoirs (Fig. 13). Total mercury levels in surficial sediments from the reservoirs were also generally similar to those in the control lakes (Table 7). In the few instances where high mercury levels were measured in sediments from the reservoirs (eg. Methyl Bay), the methylation rates in those sediments were the same as at reservoir sites where sediment mercury levels were comparable to the control lakes. These findings indicate either that mercury was not the factor limiting the rate of methylation in the lakes/reservoirs studied or that this mercury was not available to the methylating bacteria. In either case it is apparent that the fish mercury problem in lakes flooded by the Churchill River diversion was not the product of increased concentrations of total mercury. As such it will not be possible to predict the duration of the problem from the rate of depletion of total mercury in flooded material as suggested by Meister et al. (1979) and Cox et al. (1979).

While it is not possible with the results of this study to completely eliminate mercury sources as the cause of the fish mercury problem in lakes flooded by the Churchill River diversion the role of mercury additions is certainly questionable. A more likely cause of the elevated methylation rates in the reservoirs appears to have been the influx of bacterial nutrients in the inundated terrestrial soils and vegetation, as suggested by Bodaly et al. (1984). Although microbial activity was not measured, the circumstances indicate that, at most of the sites, the elevated methylation rates were the result of a direct stimulation of the growth of the methylating bacteria rather than a modification of the chemical environment by the decomposition of this material. Biological mercury methylation is strongly influenced by oxygen concentration (Ramlal 1983; Rudd et al. 1983), increasing dramatically when oxygen levels decline below about 3-4 ppm (Fagerstrom and Jernelov 1972). Reduced near-bottom oxygen concentrations were not generally found in the reservoirs (Table 3) and therefore were not primarily responsible for the elevated methylation rates. However, the depletion of oxygen at depth at the Issett L. site (Table 3) was sufficient to account for the much higher rate of methylation observed there than at the other sites along the Churchill River diversion (Table 4).

Without question, the vegetation and organic soils in the flooded zones of the reservoirs represent a source of nutrients not available in the control lakes. Inundated vegetation contributed an estimated 5.87 10^{12} g C to Southern Indian Lake (G.K. McCullough and J.M. Shay, unpublished data). This represents a loading of 2.46 10^{3} g C m^{-2} of lake area or 1.42 10^{4} g C m^{-2} of flooded area. In comparison, annual primary production in 1977 was just 55 g C m^{-2} (Hecky and Guildford 1984). Numerous studies have found that methylation rates are elevated by the addition of an organic substrate (Furutani and Rudd 1980; Wright and Hamilton 1982; Jensen and Jernelov 1969; Jernelov 1972; Bishop and Kirsch 1972). In all cases, the increased rates of methylation were accompanied by a complementary increase in microbial activity. In a series of enclosure experiments conducted at Southern Indian lake (Hecky et al. 1987 a and b) additions of Sphagnum sp. moss and peat, prairie soil, or black spruce (Picea mariana) boughs all resulted in elevated fish mercury levels.

A prediction of the duration of the fish-mercury problem

For a difference in the methylation balance in reservoirs of different ages to be indicative of an aging effect, it must be reasonable to expect similar net rates of methyl mercury production over the same type of flooded material in reservoirs of comparable age. The measurements made in lakes along the Churchill River diversion indicate that such an expectation is realistic provided the oxygen concentrations at the sites are similar. Of the four sites surveyed, the near-bottom waters were well-oxygenated at three sites (Table 3) and the seasonal mean methylation rates at the inshore stations of these locations were nearly identical (Table 4). The similarity of methylation rates at these sites is also indicated by the finding of Bodaly et al. (1984) that fish mercury levels in lakes along the diversion were proportional to the percentage increase in lake area due to flooding. This relationship could only occur if the rate of methyl mercury production per unit area of flooded material was the same at all locations.

The similarity of the seasonal mean methylation rates measured in reservoirs ranging from 8 to 42 years of age (Table 4) suggests that the fish mercury problem in lakes flooded by the Churchill River diversion will be long-lived, continuing for more than 30 years. Reservoirs older than Reindeer

L. should be studied to provide a more precise estimate of the duration of the problem. Given the apparent cause of the elevated fish mercury levels in lakes flooded by the Churchill River diversion, the time course will be determined by the rate at which organic material in the flooded zone is depleted or made unavailable to the methylating bacteria. It is unlikely that a significant amount of flooded vegetation will be flushed from the system. Hecky and McCullough (1984) estimated that 90% of the mineral sediment eroded in the 3 years following impoundment of SIL was deposited within the lake. An even higher proportion of the inundated vegetation is expected to remain in the lake owing to the generally larger particle sizes.

The most important factor determining the duration of the fish mercury problem will be the rate of decomposition of the flooded material. The considerable length of the projected duration of problem in lakes along the diversion route is likely a product of the type of vegetation that was flooded. Approximately 80% of the vegetation biomass innundated by the flooding of SIL is tree bole (G.K. McCullough, Department of Fisheries and Oceans, Freshwater Institute, Winnipeg, MB. pers. com.) which is resistant to breakdown. Anderson et al. (1978) estimated degradation times for this material of as long as 200 years. Similarly, Hodkinson (1975) predicted a half life of 57 years and a 95% life of 246 years for 7.5 cm diameter logs of _Populus balsamifera_ in an Alberta beaver pond. The refractory nature of the material was also quite apparent on our visits to Reindeer Lake, where tree trunks and wood chips litter the nearshore waters. However, the resistance of this material to decomposition may have prevented the occurrence of much higher fish mercury levels which might have developed if a more easily decayed vegetation type had been flooded.

Our conclusion is quite in contrast to that of Abernathy and Cumbie (1977) and Cox et al. (1979) who feel that the elevation of fish mercury levels in new reservoirs is transitory, declining within 3-5 years of impoundment. These estimates were derived from studies in Illinois and South Carolina, respectively, and it is apparent that they are not applicable to reservoirs in the subarctic zone. Although it is not possible to support our measurement of elevated methylation rates in Reindeer L. with measurements of elevated fish mercury levels, fish mercury levels in Stephens Lake remain unacceptably elevated 13 years after reservoir formation. Similarly, Lodenius et al. (1983) found that fish mercury levels in a Finnish reservoir remained elevated 14 years after formation. The relative persistence of the mercury problem in northern reservoirs will be due, at least in part, to the climatic difference, with decomposition of flooded material proceding more slowly at lower temperature. The type and amount of vegetation flooded will also be important but it is not possible to evaluate the significance of any differences with the present state of knowledge. We plan to extend the use of this method to other reservoirs to study the effects of different vegetation types and loading rates on the methylation balance.

It is becoming increasingly clear that high fish mercury levels are to be expected in new reservoirs and must be considered in their planning. Accurate predictions of the severity and duration of resulting mercury problems will be fundamental to the planning process. Comparisons of conditions in reservoirs of different age are presently the only foundation for such predictions and then primarily for estimates of duration. This study has shown that the methylation balance has been shifted in favour of net methyl mercury production in the flooded zones of a number of northern Manitoba reservoirs and that comparisons of the methylation balance among reservoirs of different age should be useful to predict the evolution of the fish mercury problem in lakes flooded by the Churchill River diversion. These comparisons indicate that unacceptably high fish mercury levels may be much more persistent in northern reservoirs, lasting for decades, than would be predicted from investigations in more temperate regions. It is also apparent, from the results of this and related studies of lakes along the diversion, that the fish mercury problem in new reservoirs can no longer simply be considered a mercury source problem and may not be the result of an influx of mercury at all. Comparisons of methylating activity among reservoirs also appear to represent a good alternative to comparisons of fish mercury levels as a predictive tool because the interpretive problems associated with differences in fish species, size, and age and with the dependence of fish mercury levels on the extent of flooding are eliminated.

Acknowledgements

Special thanks to Maragret Treble for her able assistance in the field and lab. We would also like to thank M. Walker and the pilots of Parsons Airways (Northern) Ltd. for their assistance with field sampling. The design of this study benefitted greatly from early discussions with R. Hecky and J. Rudd. The total mercury analyses were performed by R. Hunt and the methyl mercury analyses were done by B. Grift, both of the Freshwater Institute. Dr. J. Rudd provided the labelled methyl mercury and access to equipment.

This study was conducted under DSS contract file numbers 01SF.FP430-3-S096 and 01SF.FP430-5-S9015 with funding provided by the Canada-Manitoba Agreement on the Study and Monitoring of Mercury in the Churchill River diversion and the Department of Fisheries and Oceans.

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Table 1. Histories of reservoirs included in the survey. Sources: McCullough (1981), Brown (1974), Water Resources Branch (1974), Vitkin (1979), Water Survey of Canada (1939-1983).

Reservoir	Year(s) Flooded	Mean depth of flooding (m)	Range of level fluctuation (m)		Area of land flooded (km^2)	% increase in lake area
			Mean	Max.		
Southern Indian Lake	1976	3.0	0.9	1.3	414	21
Notigi Reservoir					431	282
Notigi Lake	1974-76	15.2	2.2	3.1	43	387
Issett Lake	1976	7.6	0.9	1.3	70	800
Stephens Lake	1970	NA	2.6	3.1	NA	NA
Sipiwesk Lake	1960-65	3.0[a]	2.3	3.1	>26[b]	>7.5[c]
Reindeer Lake	1939-42	2.1	0.6	1.2	<26	0.5

---

[a] Water Resources Branch (1974) states 2 m but Water Survey of Canada records (1960 to 1965) indicate 3 m is a better approximation

[b] 26 km^2 based on a 2 m level increase (Water Resources Branch 1974)

[c] 7.5% increase in lake area based on a 2 m level increase (Water Resources Branch 1974)

NA data not available

Table 2. Key to sampling sites, date of sampling, and station depth in the 1984 survey of northern Manitoba reservoirs.

Lake/Reservoir	Site	Station		Sampling date and station depth (m)				
	Name	Number	Type	June 26	July 10	August 10	August 23	September 28
Southern Indian Lake	Region 5	1	Flooded-inshore	3.2	4.0	2.4	3.0	1.9
		2	Flooded-inshore	3.2	4.2	2.3	3.0	3.2
		3	Offshore	6.5	8.0	4.0	5.7	5.8
		4	Flooded-inshore			2.4		
		5	Flooded-inshore			1.7		
		6	Offshore			3.5		
				July 3	July 12	August 3	August 21	September 25
	Methyl Bay	1	Flooded-inshore	2.0	2.0	1.7	1.9	1.6
		2	Flooded-inshore	2.2	1.2	1.4	0.8	1.7
		3	Offshore	4.2	4.7	4.5	6.3	6.6
		4	Flooded-inshore			1.0		
				June 27	July 11	August 2	August 24	September 26
Notigi Reservoir	Issett Lake	1	Flooded-inshore	6.2	2.9	6.5	5.3	2.0
		2	Flooded-inshore	7.0	4.0	6.2	6.2	6.8
		3	Flooded-offshore	7.0	6.5	6.7	7.0	7.5
		4	Flooded-inshore			1.2		
		5	Flooded-inshore			4.6		
		6	Flooded-offshore			4.0		
				June 27	July 11	August 9	August 24	September 26
	West Notigi Lake	1	Flooded-inshore	3.8	2.6	2.0	2.4	2.9
		2	Flooded-inshore	6.0	5.0	3.0	2.0	3.8
		3	Flooded-offshore	9.0	12.0	11.5	7.8	13.8
		4	Flooded-inshore			1.0		
		5	Flooded-inshore			5.6		
		6	Flooded-offshore			6.3		
				June 25	July 9	August 10	August 22	September 27
Stephens Lake		1	Flooded-inshore	3.7	2.7	1.9	0.8	1.2
		2	Flooded-inshore	3.8	3.0	3.5	0.7	3.0
		3	Flooded-offshore	20.0	9.0	11.0	10.5	7.8
		4	Flooded-inshore			1.6		
		5	Flooded-inshore			2.0		
		6	Flooded-offshore			4.3		
				June 25	July 9	August 9	August 22	September 27
Sipiwesk Lake		1	Inshore	2.0	1.4	1.0	1.4	1.0
		2	Inshore	1.4	1.4	0.8	1.1	0.4
		3	Offshore	4.5	5.5	3.8	4.8	5.5
		4	Inshore			0.8		
		5	Inshore			2.0		
		6	Offshore			5.3		
				June 26	July 10	August 08	August 23	September 28
Reindeer Lake		1	Flooded-inshore	1.1	2.4	0.7	1.2	1.8
		2	Flooded-inshore	1.3	N/A	1.1	1.2	1.9
		3	Offshore	5.2	5.0	5.0	5.3	4.3
		4	Flooded-inshore			0.7		
		5	Flooded-inshore			1.3		
		6	Offshore			4.0		
				June 27	July 11	August 9	August 24	September 26
East Mynarski Lake		1	Inshore	3.4	2.8	1.9	0.9	1.3
		2	Offshore	3.5	4.0	4.4	5.0	4.7
		3	Inshore			2.0		
		4	Inshore			0.8		
		5	Inshore			0.8		
				June 26	July 10	August 11	August 23	September 28
Granville Lake		1	Inshore	1.6	1.1	1.0	1.8	0.8
		2	Offshore	7.0	7.2	6.6	6.6	6.0
		3	Inshore			1.1		
		4	Offshore			6.9		
		5	Inshore			0.8		
		6	Inshore			0.8		

Table 3. Time-weighted mean Secchi disc visibility and time-weighted means of selected physical and chemical characteristics of the near-bottom waters at the inshore and offshore stations at each of the lake and reservoir sites surveyed in 1984 (see Fig. 1 for locations). Inshore and offshore stations are as defined in Table 2.

Site	Location	Secchi (m)	Conductivity (uS cm^{-1})	pH	Temperature (^{o}C)	Dissolved oxygen (ppm)
Granville L.	Inshore	1.44	70	7.4	17.22	9.1
	Offshore		70	7.1	16.72	8.4
East Mynarski L.	Inshore	1.60	133	7.9	16.69	9.4
	Offshore		133	7.9	16.85	8.7
SIL Region 5	Inshore	2.29	115	7.5	13.17	9.5
	Offshore		114	7.5	13.07	9.1
Methyl Bay	Inshore	0.85	81	7.4	16.29	9.1
	Offshore		80	7.2	15.57	8.5
West Notigi L.	Inshore	1.34	96	7.2	16.51	8.6
	Offshore		96	6.9	16.13	7.1
Issett L.	Inshore	2.20	139	6.6	13.65	3.1
	Offshore		150	6.5	11.67	2.5
Stephens L.	Inshore	0.49	204	7.7	14.35	8.9
	Offshore		206	7.8	15.61	8.9
Sipiwesk L.	Inshore	0.51	163	8.1	17.23	9.0
	Offshore		163	7.8	16.55	8.3
Reindeer L.	Inshore	>5.20	38	7.1	15.41	9.8
	Offshore		38	6.9	14.86	9.5

Table 4. Seasonal mean (time-weighted) specific rates of mercury methylation and methylmercury demethylation and the ratio of these rates (M/D) at the lake and reservoir sites surveyed in 1984 (see Fig. 1 for locations). Inshore and offshore stations are as defined in Table 2.

Site	Station location	% added Hg g^{-1} h^{-1} Methylation	Demethylation	M/D
Granville L.	inshore	0.008	0.185	0.043
	offshore	0.001	0.132	0.009
East Mynarski L.	inshore	0.011	0.273	0.038
	offshore	0.04	0.518	0.077
Methyl Bay	inshore	0.016	0.096	0.162
	offshore	<0.001	0.086	0.004
SIL Region 5	inshore	0.02	0.272	0.072
	offshore	0.01	0.158	0.061
West Notigi L.	inshore	0.02	0.132	0.155
	offshore	0.027	0.122	0.222
Issett L.	inshore	0.079	0.143	0.551
	offshore	0.088	0.099	0.892
Stephens L.	inshore	0.017	0.147	0.113
	offshore	0.053	0.146	0.362
Sipiwesk L.	inshore	0.003	0.181	0.015
	offshore	0.002	0.115	0.017
Reindeer L.	inshore	0.019	0.097	0.199
	offshore	0.022	0.109	0.203

Table 5. Mean specific methylation and demethylation rates and coefficients of variation (C of V) at the inshore stations at each of the sites sampled on the third survey (August 2 to 11) in 1984 (see Fig. 1 for locations). Inshore stations are as defined in Table 2.

Site	% added Hg g^{-1} h^{-1} Methylation Mean	Methylation C of V (%)	Demethylation Mean	Demethylation C of V (%)
Granville L.	0.027	44.1	0.099	22.1
East Mynarski L.	0.026	31.6	0.309	44.0
SIL Region 5	0.068	31.4	0.143	11.8
Methyl Bay	0.059	79.6	0.130	23.3
West Notigi L.	0.057	45.9	0.143	19.3
Issett L.	0.087	56.7	0.184	22.6
Stephens L.	0.069	52.1	0.149	29.8
Sipiwesk L.	0.011	65.2	0.153	27.0
Reindeer L.	0.057	45.9	0.087	22.5

Table 6. Descriptive statistics for the cumulative frequency distributions (Fig. 10) of specific methylation rates at the control stations and at the flooded inshore stations measured during the third survey in 1984. Calculations were as described in Huntsberger and Billingsly (1977).

	Lower quartile	Median	Upper quartile	Trimean	Mean
Controls	0.002	0.002	0.05	0.014	0.028
Reservoirs	0.003	0.064	0.123	0.064	0.073

Table 7. Total mercury concentrations (ng (g sed. dry wt)$^{-1}$) in sediments from the inshore and offshore stations at the lake and reservoir sites (see Fig. 1 for locations) on the first (June 25 to July 3) and fourth (August 21 to 24) surveys in 1984. Values for the inshore stations represent the mean concentration in two samples.

	Total Hg concentration (ng (g sed. dry wt)$^{-1}$)			
	Survey 1		Survey 4	
Site	Inshore	Offshore	Inshore	Offshore
Granville L.	55.9	157.5	72.9	38.8
East Mynarski L.	115.0	107.8	54.4	180.6
SIL Region 5	165.9	106.2	119.8	64.4
Methyl Bay	1476.9	418.0	393.3	36.6
West Notigi L.	217.5	125.0	114.9	161.7
Issett L.	277.6	376.5	143.4	84.4
Stephens L.	84.4	51.6	108.1	46.2
Sipiwesk L.	73.3	94.3	96.5	122.9
Reindeer L.	205.8	108.5	183.1	144.7

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12. Seasonal mean (time-weighted) total mercury concentration in plankton collected in a 53 um mesh-size net.

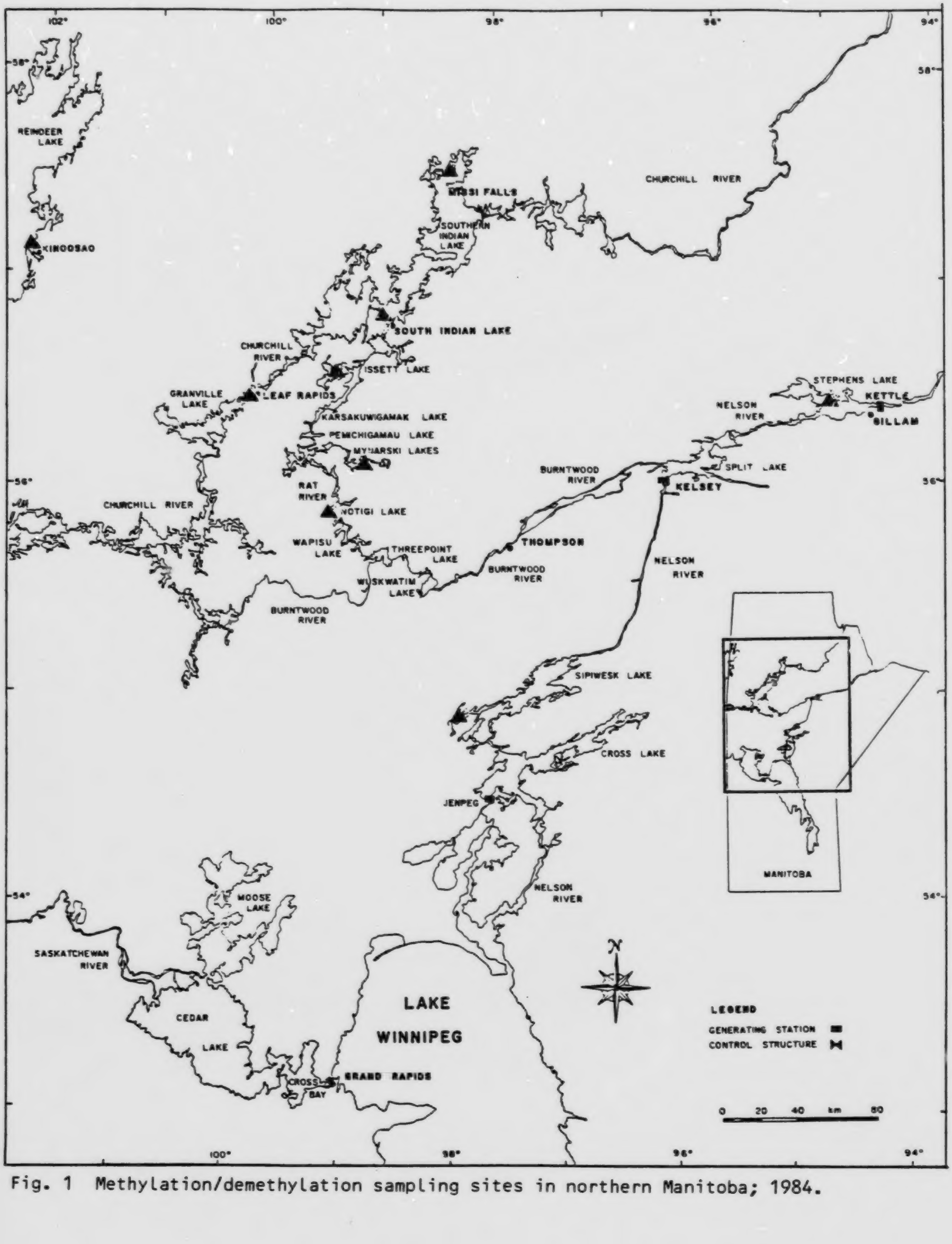

102°
100°
98°
96°
94°
58°
56°
54°
REINDEER LAKE
KINOOSAO
MISSI FALLS
CHURCHILL RIVER
SOUTHERN INDIAN LAKE
SOUTH INDIAN LAKE
CHURCHILL RIVER
ISSETT LAKE
GRANVILLE LAKE
LEAF RAPIDS
KARSAKUWIGAMAK LAKE
PEMICHIGAMAU LAKE
MYNARSKI LAKES
STEPHENS LAKE
KETTLE
GILLAM
NELSON RIVER
SPLIT LAKE
KELSEY
BURNTWOOD RIVER
RAT RIVER
NOTIGI LAKE
CHURCHILL RIVER
WAPISU LAKE
THREEPOINT LAKE
THOMPSON
WUSKWATIM LAKE
BURNTWOOD RIVER
BURNTWOOD RIVER
NELSON RIVER
SIPIWESK LAKE
CROSS LAKE
JENPEG
MANITOBA
MOOSE LAKE
NELSON RIVER
SASKATCHEWAN RIVER
N
LAKE WINNIPEG
CEDAR LAKE
CROSS BAY
GRAND RAPIDS
LEGEND
GENERATING STATION
CONTROL STRUCTURE
0 20 40 km 80
100°
98°
96°
94°

Fig. 1 Methylation/demethylation sampling sites in northern Manitoba; 1984.

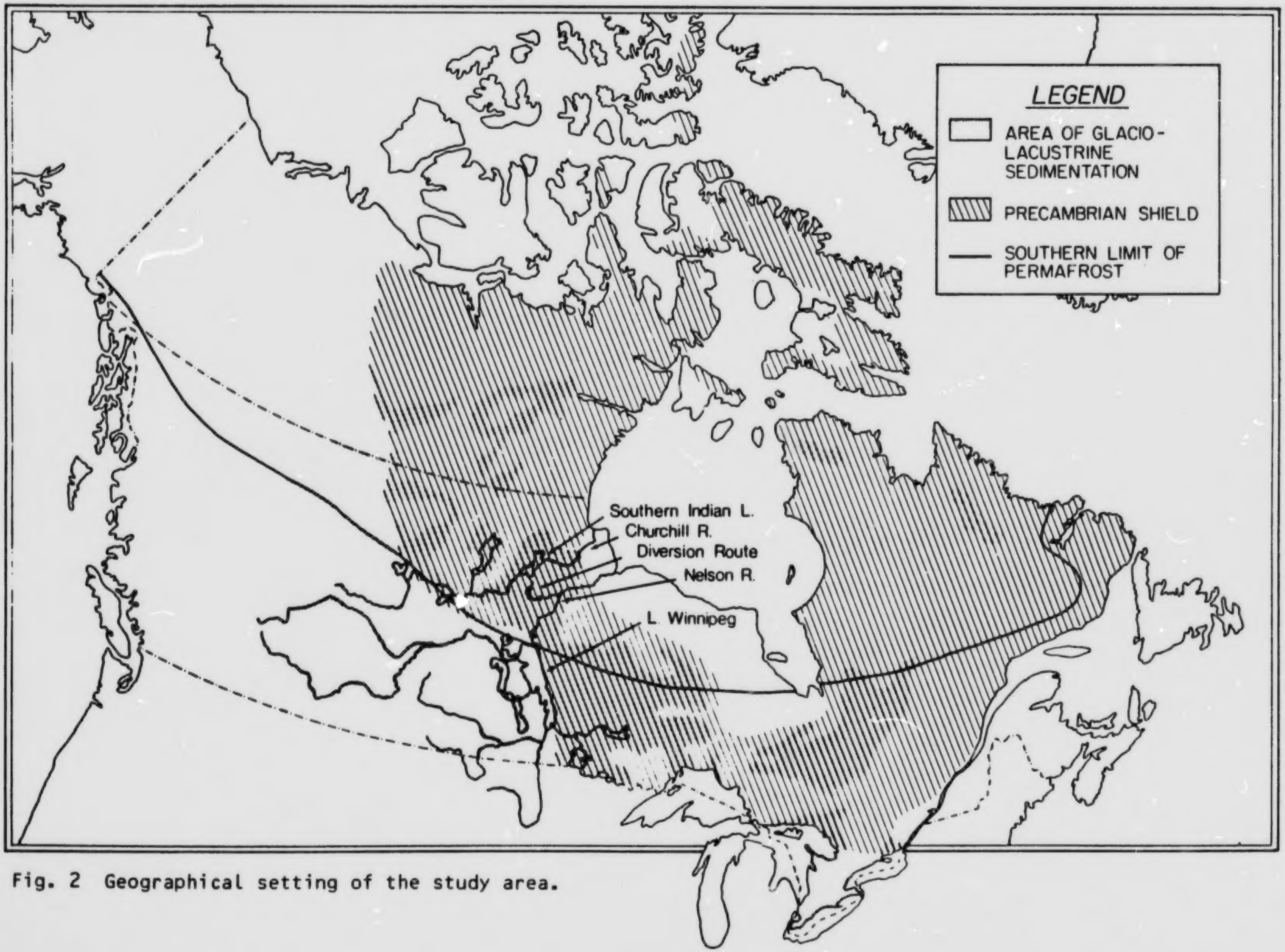
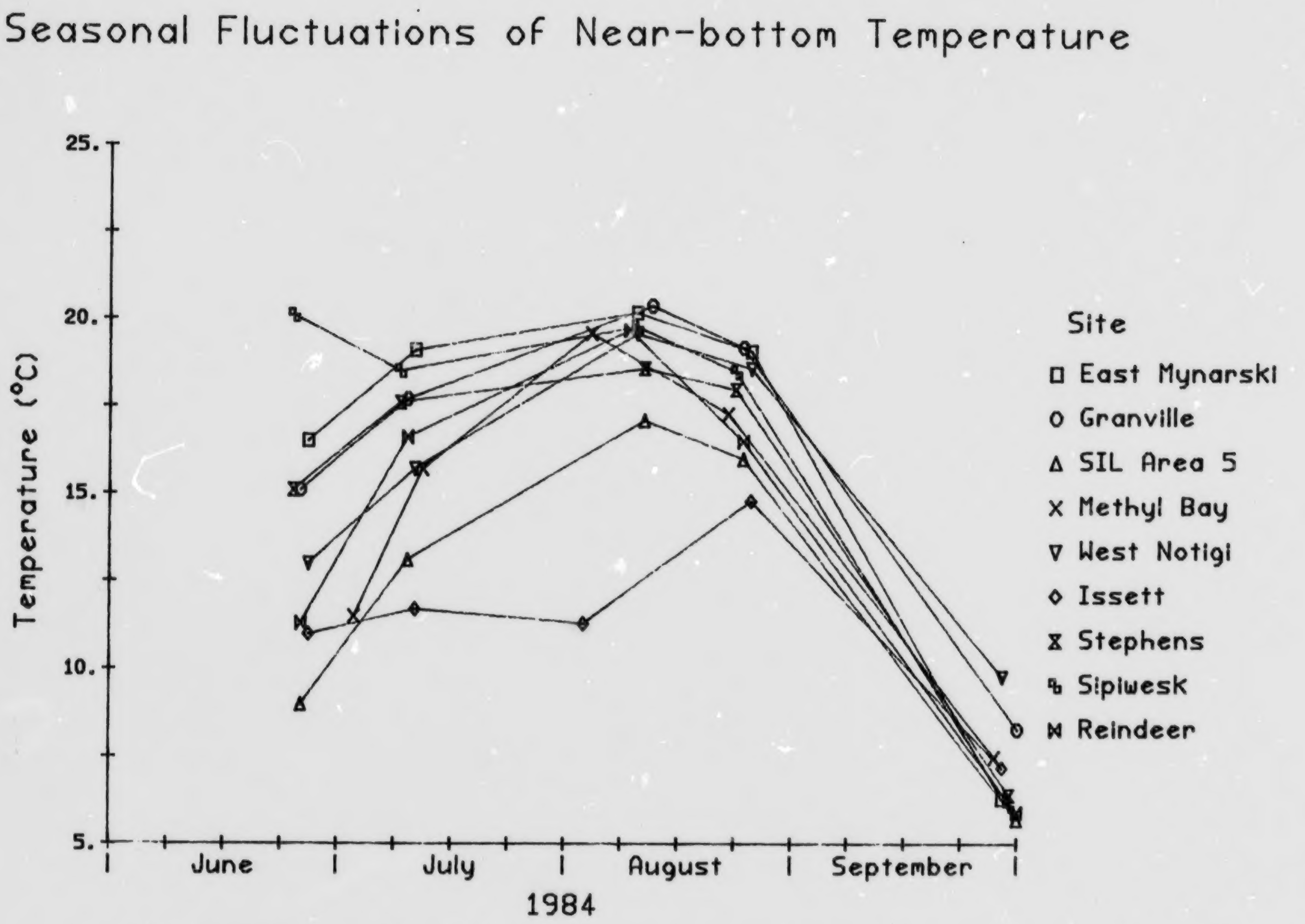

LEGEND
AREA OF GLACIO-LACUSTRINE SEDIMENTATION
PRECAMBRIAN SHIELD
SOUTHERN LIMIT OF PERMAFROST
Southern Indian L.
Churchill R.
Diversion Route
Nelson R.
L. Winnipeg

Fig. 2 Geographical setting of the study area.



Seasonal Fluctuations of Near-bottom Temperature

Fig. 3 Seasonal fluctuation of near-bottom temperature at the lake and reservoir sites surveyed in 1984. Sampling dates are listed in Table 2.

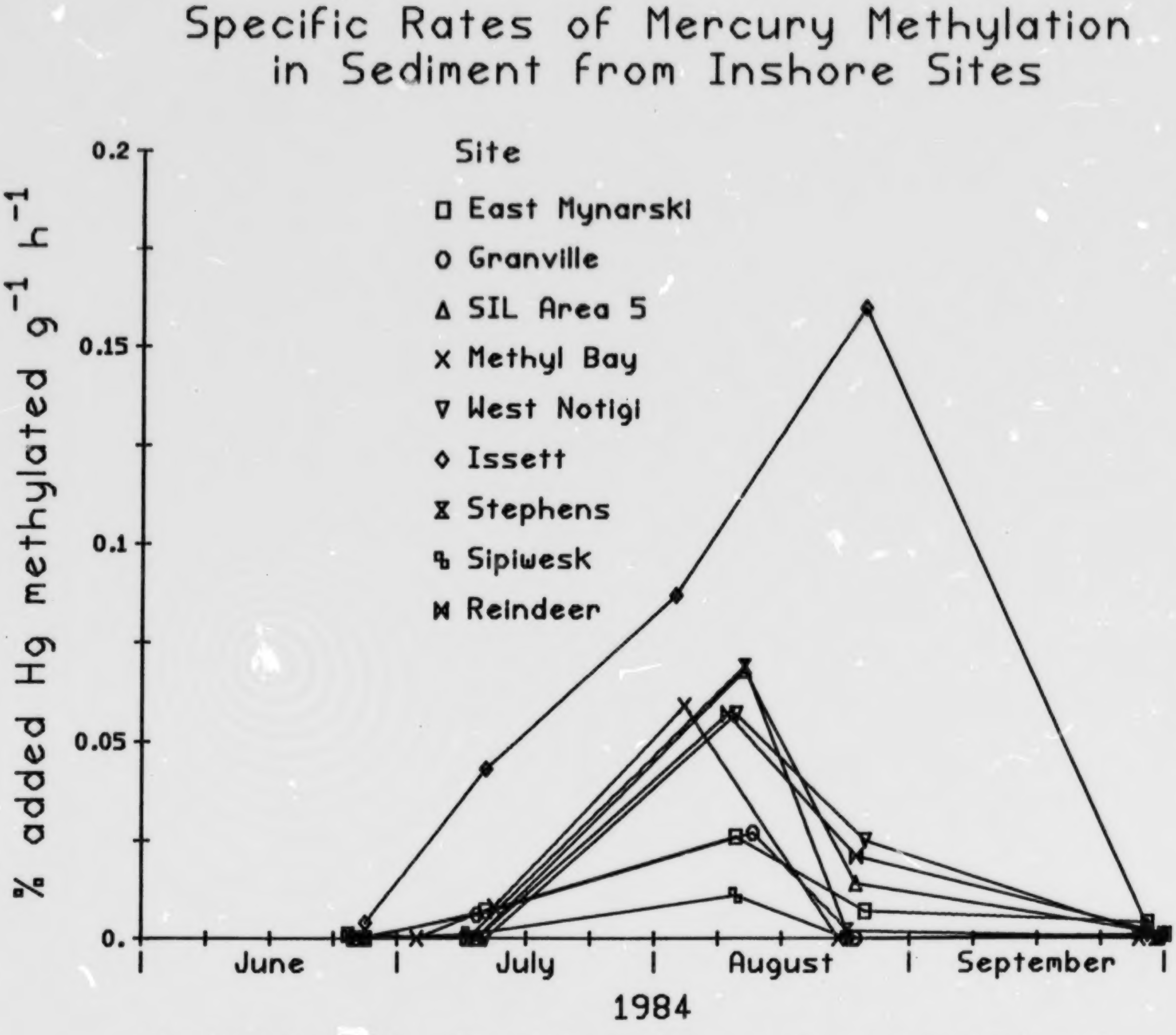


Fig. 4 Seasonal fluctuation of the specific rate of mercury methylation in sediments from the inshore stations.

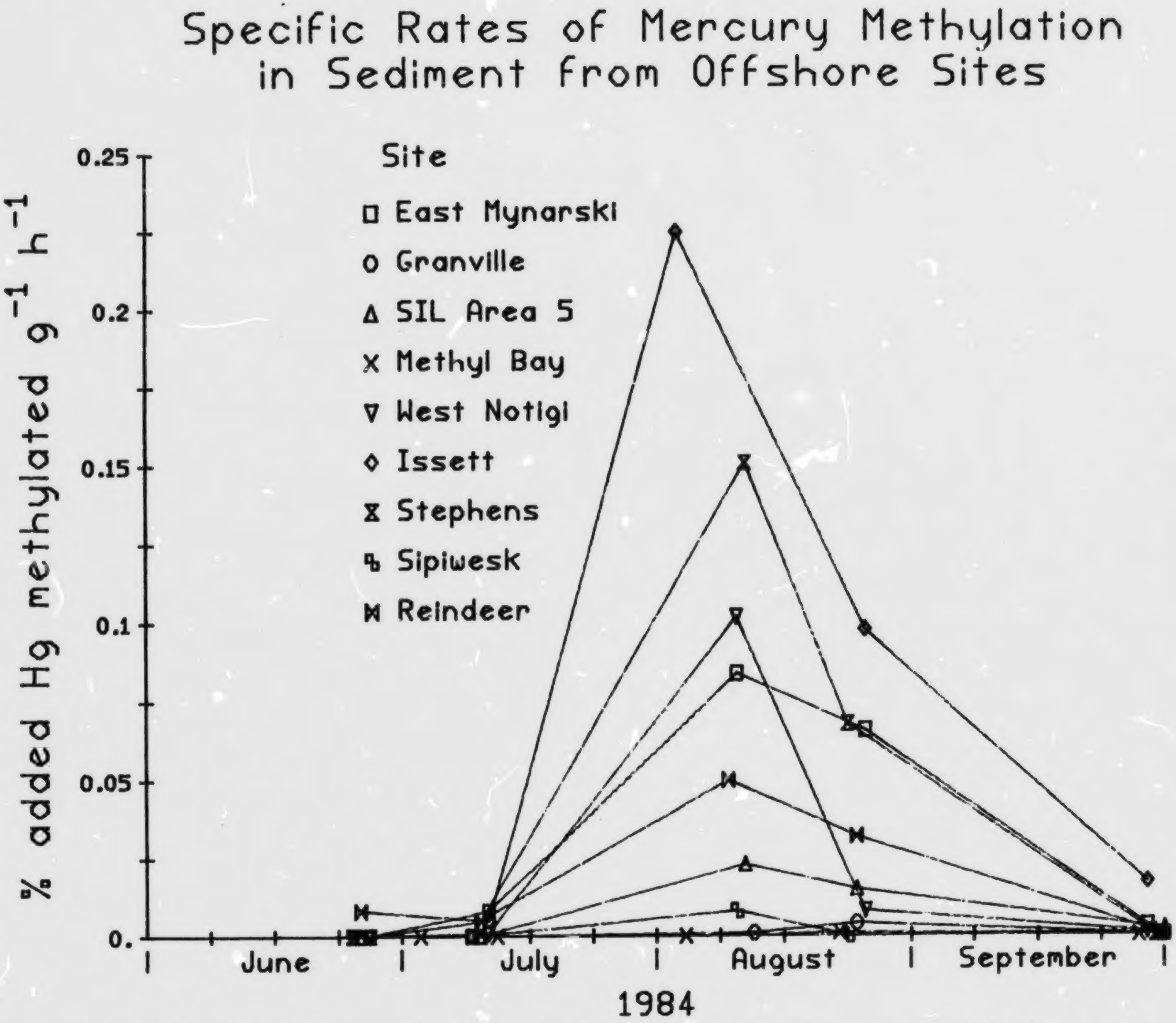
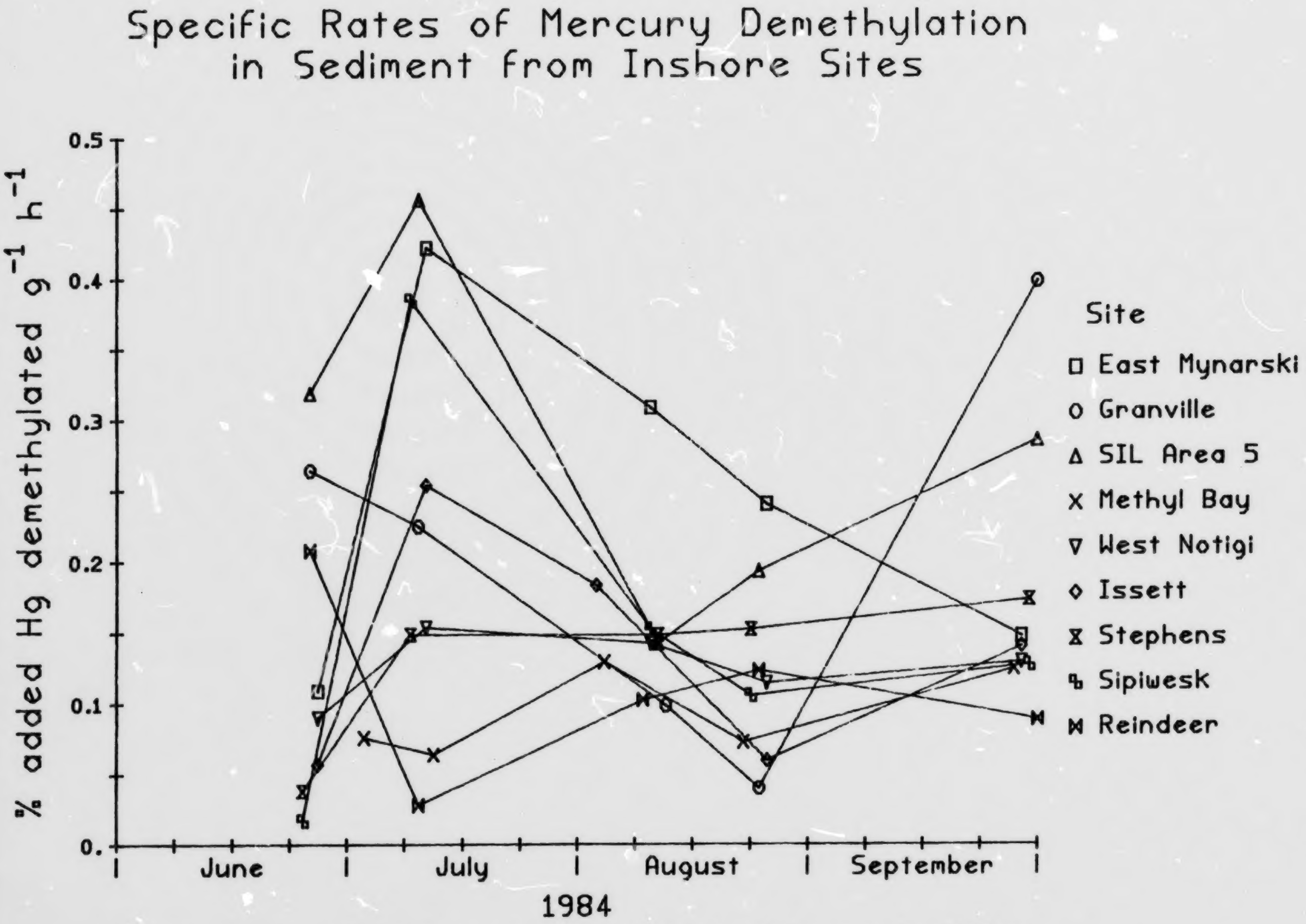


Fig. 5 Seasonal fluctuation of the specific rate of mercury methylation in sediments from the offshore stations.



Specific Rates of Mercury Demethylation in Sediment from Inshore Sites

Fig. 6 Seasonal fluctuation in the specific rate of methylmercury degradation (demethylation) in sediments from the inshore sites.

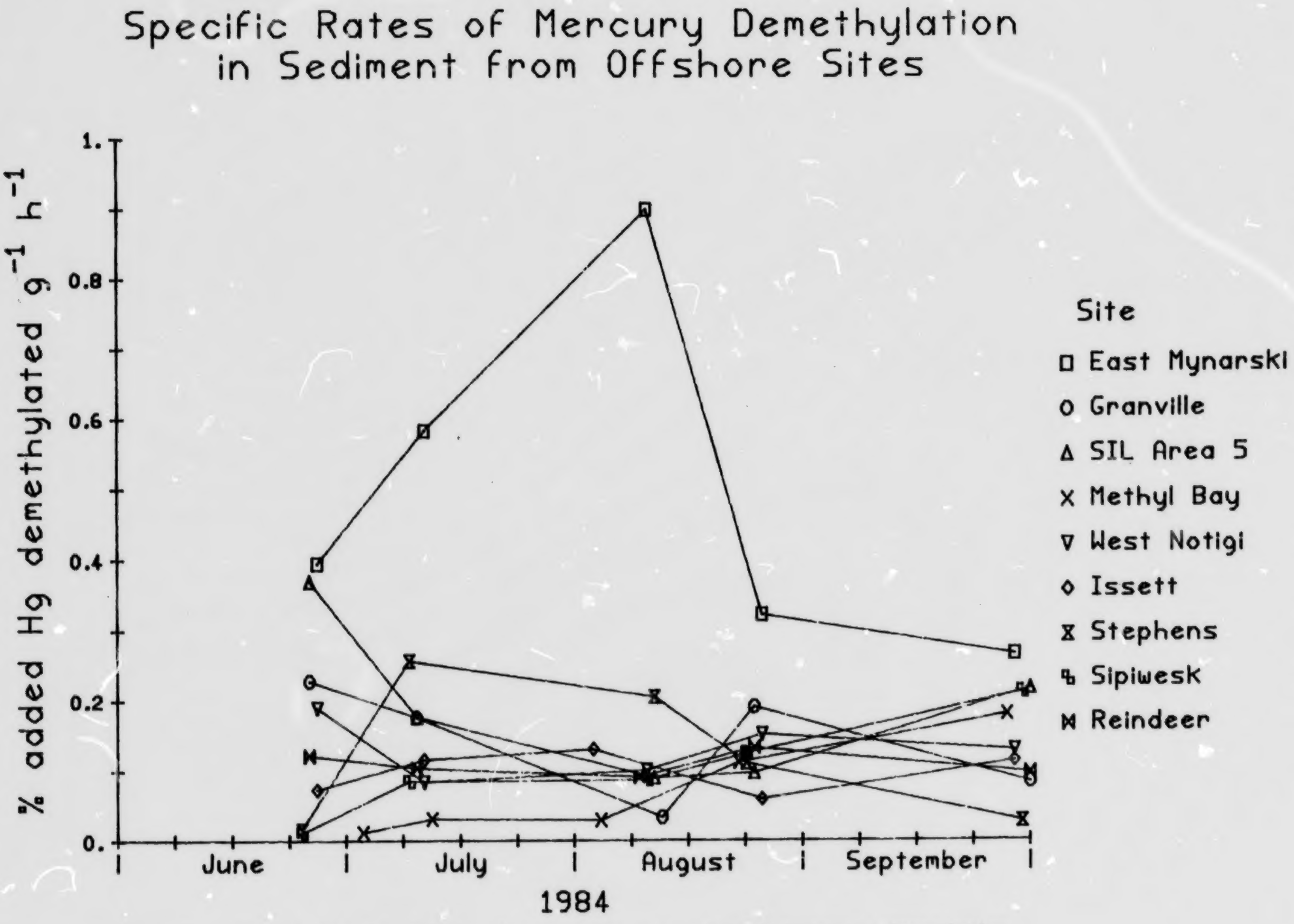


Fig. 7 Seasonal fluctuation in the specific rate of methylmercury degradation (demethylation) in sediments from the offshore sites.

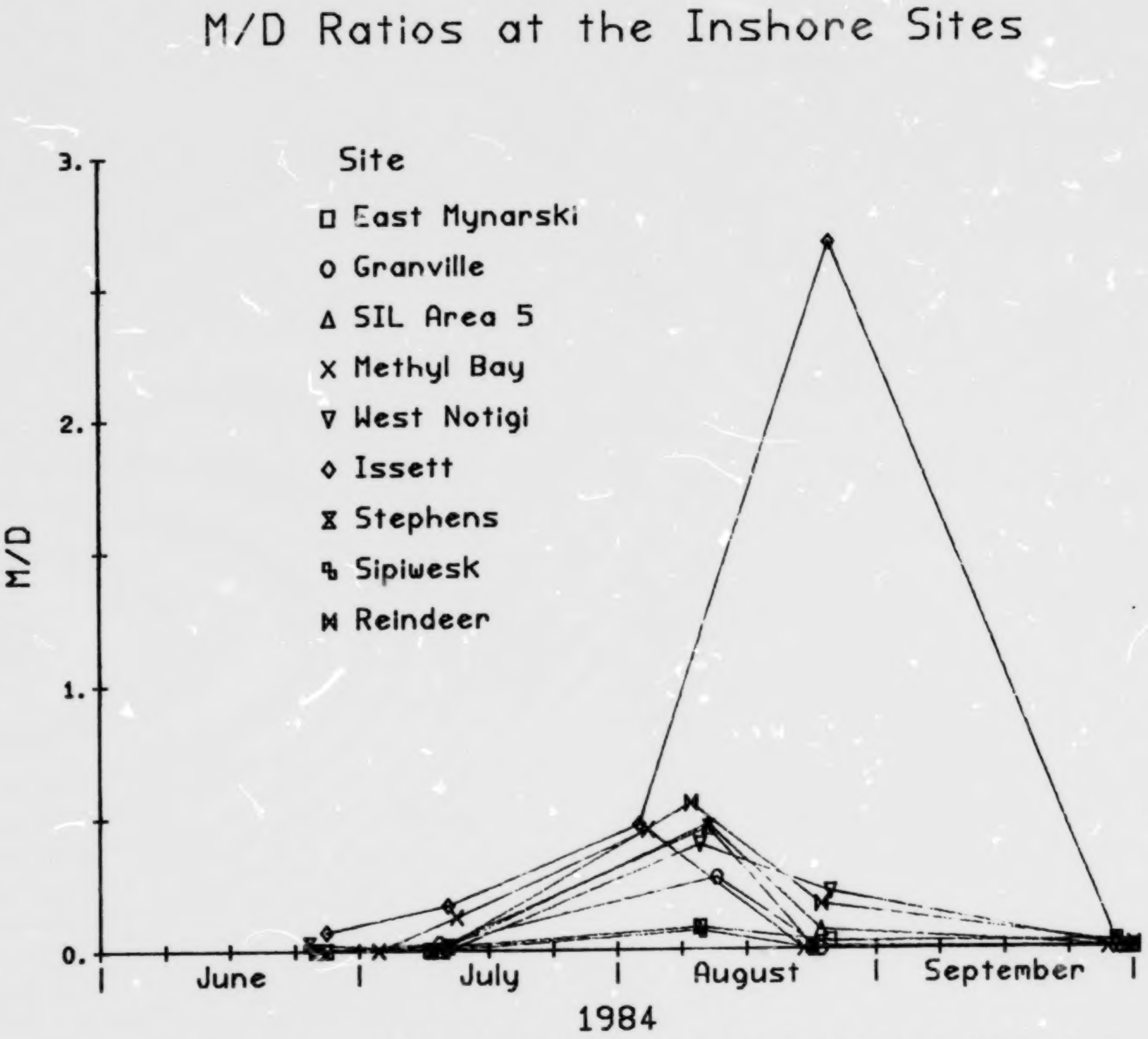


Fig. 8 Seasonal fluctuation in the methylation balance (M/D) in sediments from the inshore stations.

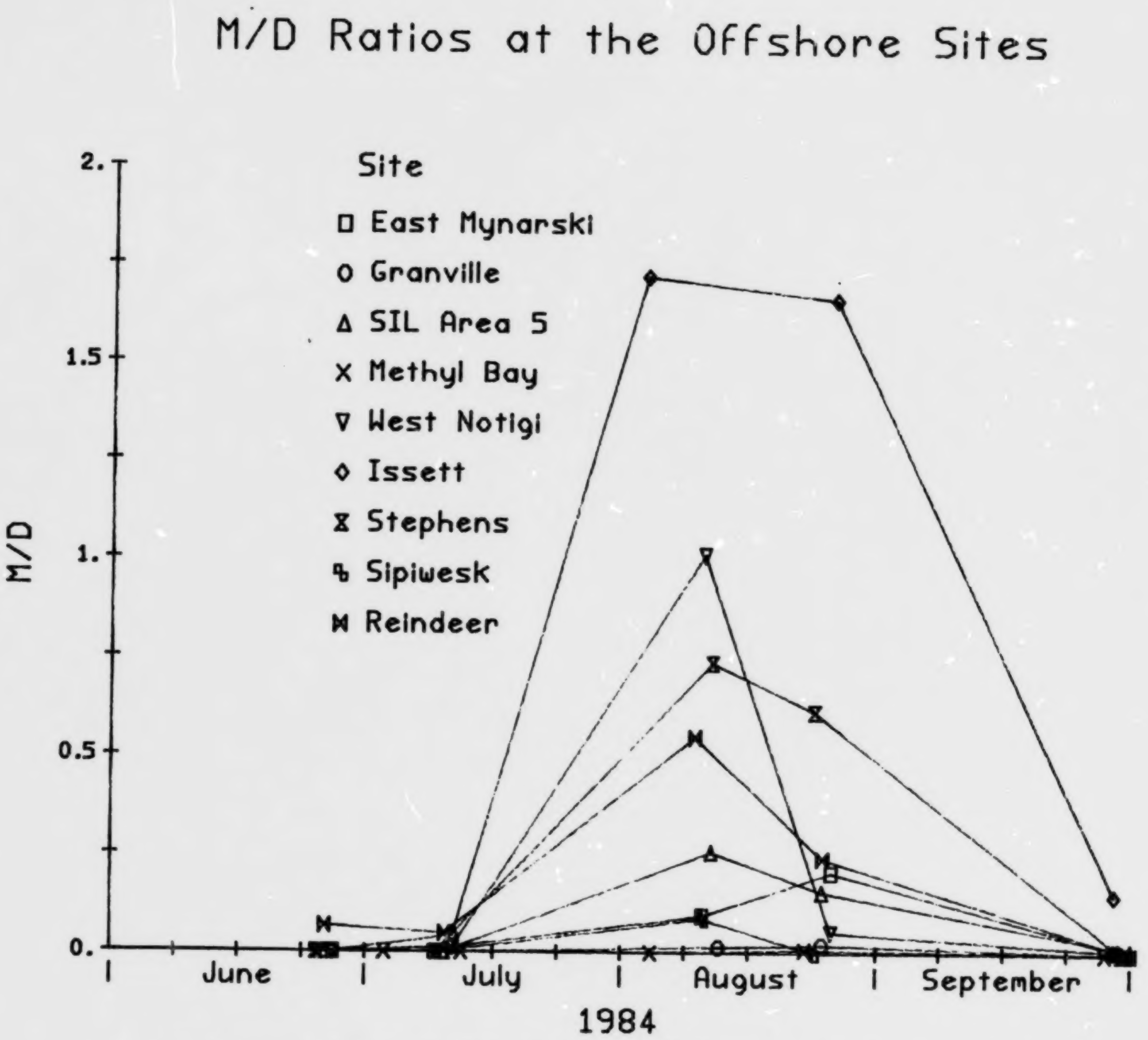


Fig. 9 Seasonal fluctuation in the methylation balance (M/D) in sediments from the offshore stations.

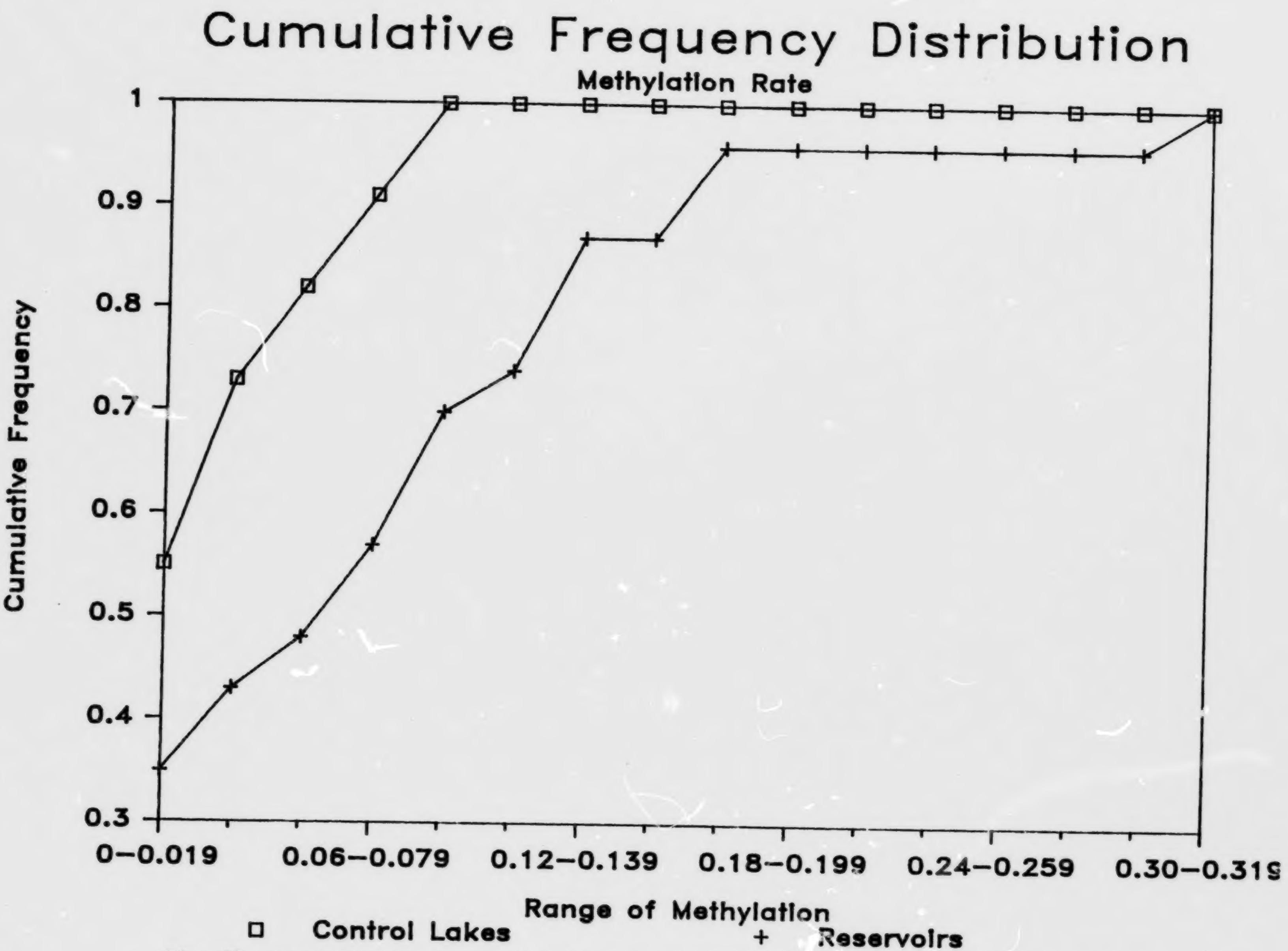


Fig. 10 Cumulative frequency distribution of the specific methylation rate in sediments from the inshore stations in the control lakes (Granville and East Mynarski) and in the reservoirs (SIL Region 5 and Methyl Bay, Notigi Lake, Stephens Lake, Reindeer Lake).

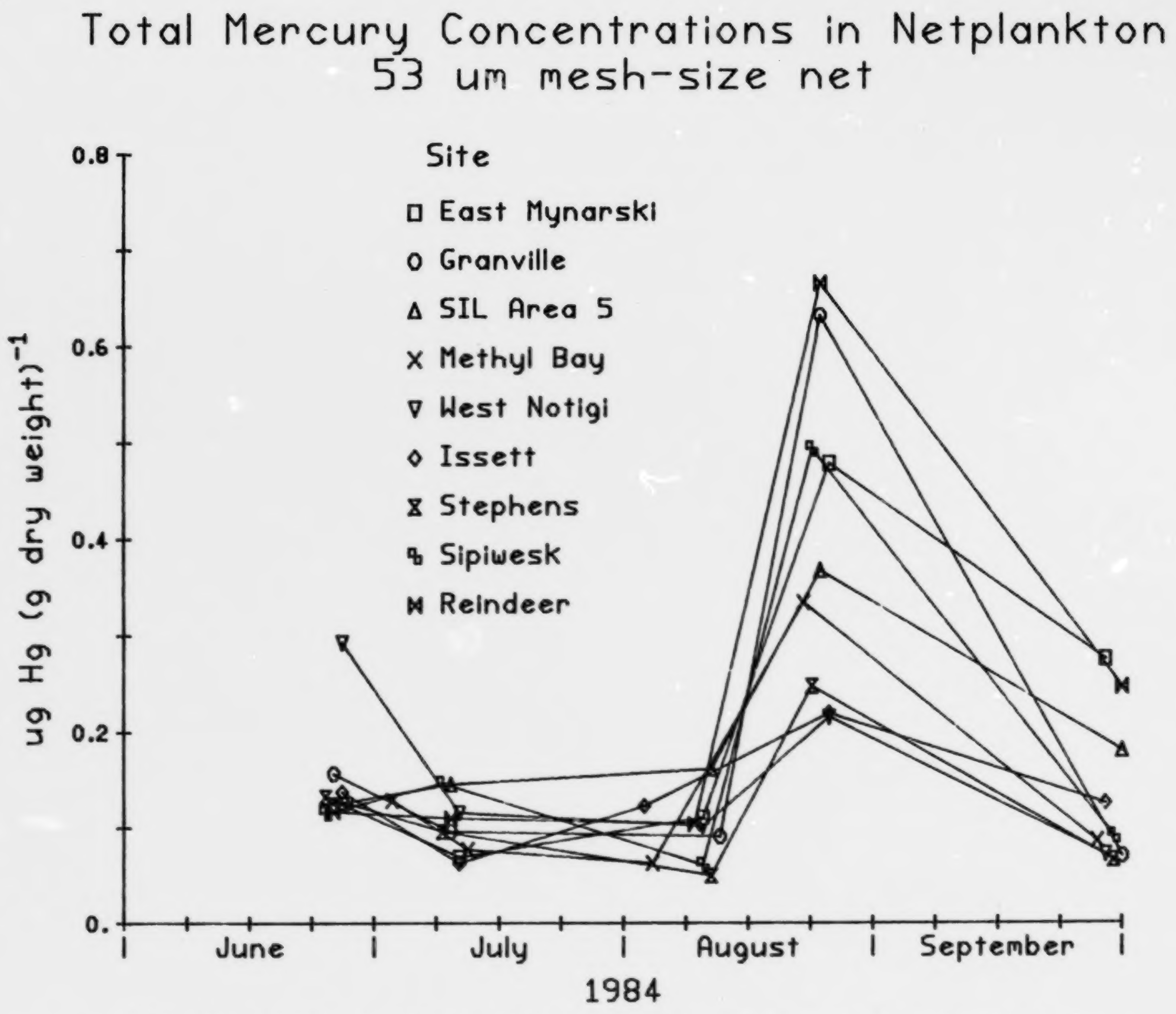


Fig. 11 Seasonal fluctuation in the total mercury concentration in plankton collected in a 53 um mesh-size net.

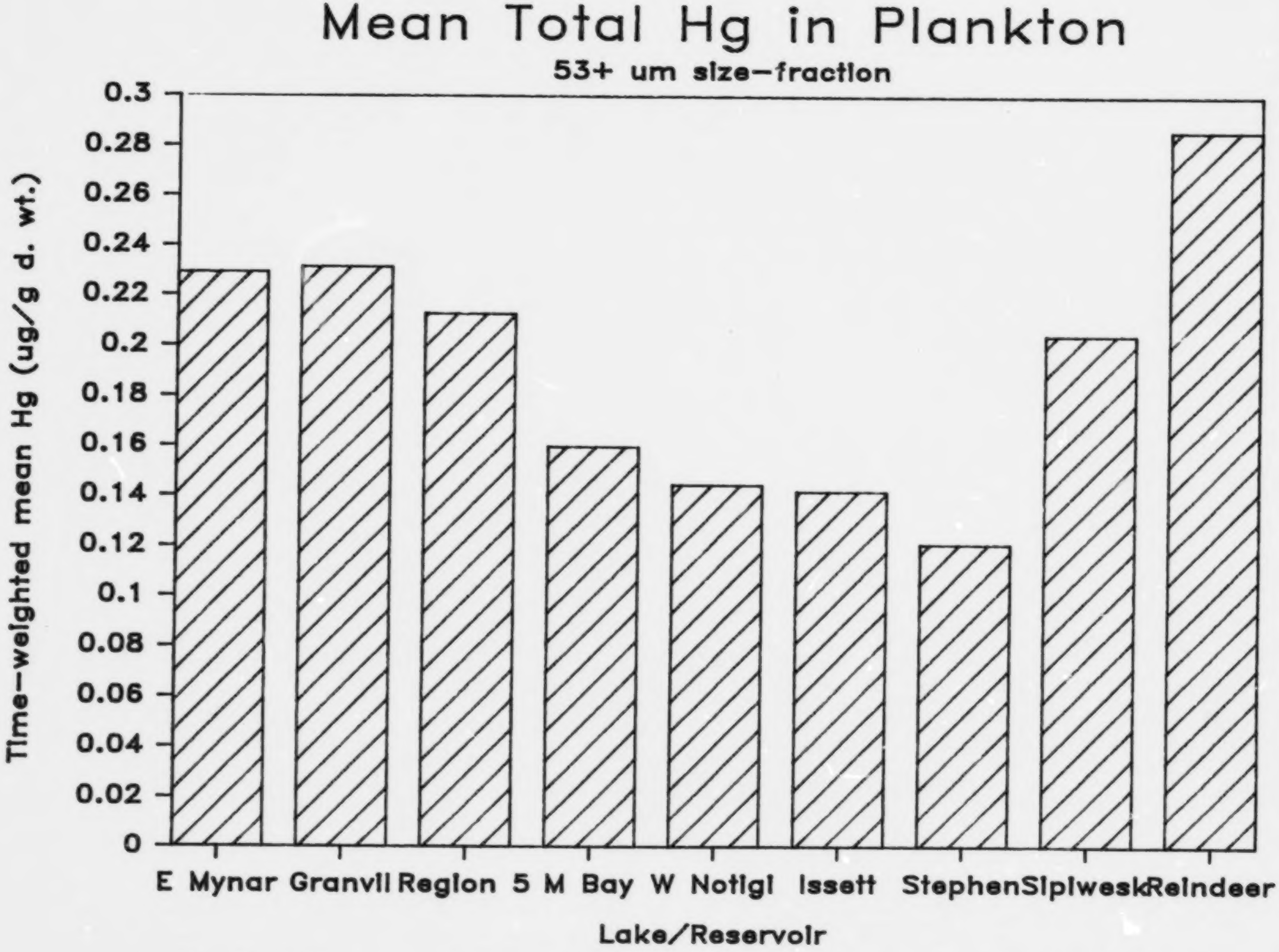


Fig. 12 Seasonal mean (time-weighted) total mercury concentration in plankton collected in a 53 um mesh-size net.

Appendix 4: Measurements of rates of production and degradation of methyl mercury and concentrations of total mercury in Southern Indian Lake, Cedar Lake, and Granville Lake, Manitoba; Results of a survey conducted in July and August, 1985

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SPECIAL NOTE

This report was prepared for the Canada-Manitoba Agreement on the Study and Monitoring of Mercury in the Churchill River Diversion. Any conclusions, recommendations, or opinions expressed herein are those of the author and are not necessarily the same as those of the Steering Committee for this Agreement.

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Introduction

Unacceptably high mercury levels in fish from lakes flooded by the Churchill River diversion were first recorded in late 1978. Bodaly and Hecky (1979) demonstrated that these high levels developed coincidently with flooding in the two lakes for which pre- and post-impoundment fish mercury data were available. A major objective of subsequent studies under the Canada-Manitoba Agreement on the study and monitoring of mercury in the Churchill River diversion has been to establish a time-course for the mercury problem in Southern Indian Lake (SIL) and Notigi Reservoir. With no clear trend in fish mercury levels since their initial increase (Bodaly et al. 1984) and in the absence of time course data from older reservoirs, an answer to this question has not been forthcoming. However, recent studies have indicated that the evolution of the problem might be predicted by examining the balance between mercury methylation and demethylation in reservoirs of different ages (Appendix 3).

A survey of 9 reservoirs and lakes in northern Manitoba was undertaken in the summer of 1984 to examine the effect of reservoir formation and duration of impoundment on the methylation balance and on mercury availability (Appendix 3). In all of the flooded areas examined, the specific methylation rate and the ratio of specific methylation to specific demethylation were markedly higher than in the unflooded reference lakes. The similarity of methylation rates in reservoirs ranging from 8 to 42 years of age indicated that the fish mercury problem in SIL and Notigi Reservoir might be long-lived.

The estimate of the duration of the problem derived from the 1984 survey necessarily assumes that the higher methylating activity in the reservoirs represents an increase from that prior to impoundment and that the methylation balance in the reservoirs will eventually return to pre-flooding levels, accompanied by a decrease in fish mercury concentrations. The purpose of the present study is to assess the validity of this assumption by determining the methylation balance of flooded material in Cedar Lake, Manitoba where high fish mercury levels were noted following impoundment but subsequently declined (Fig. 1). The Cedar Lake data can be compared to that for flooded material in SIL where fish mercury levels remain elevated, and for materials in the nearshore region of Granville Lake, an unflooded reference lake. Mercury levels in the water column, as indicated by the concentration in net plankton, and in the sediments were also measured.

Description of the study area

Cedar Lake (latitude 53° 18'N, longitude 100° 04'W; Fig. 2), was flooded in 1964 and 1965 to provide storage for the Grand Rapids Hydroelectric Generating Station located at the east side of Cross Bay where the Saskatchewan River leaves the lake. Manitoba Hydro regulates the lake level between the long term pre-impoundment mean of 253.5 m above sea level (ASL) and 257 m ASL. The lake is drawn down over winter with minimum pool level usually recorded in April and maximum levels in mid to late summer (Green and Derksen 1980). Flooding increased the area of the lake (exclusive of Cross Bay) from 1352 km^2 to about 2400 km^2. The majority of the approximately 1000 km^2 of flooded area is located over the Saskatchewan River delta on the west side of Cedar Lake (Water Resources Branch 1974). All samples were collected near (within 0.5 km) the levee of the Saskatchewan River (Fig. 2).

The mean water level of Southern Indian Lake (Fig. 2; latitude 57°N, longitude 99°W) was raised 3 m in 1976 to facilitate the southward diversion of about 75% of the Churchill River flow to supply hydroelectric developments on the lower Nelson River (Newbury et al. 1984). All samples were collected from a small isolated bay, hereafter called Methyl Bay (Fig. 2). This is the same sampling location used in 1984 (Appendix 3) and Ramlai et al. (1986 a and b).

Granville Lake (Fig. 2) is a riverine lake on the Churchill River immediately upstream from SIL which has not been affected by the flooding. Samples were collected from approximately the same location (Fig. 2) used in 1984 (Appendix 3) and Ramlal et al. (1986 a and b).

Methods

Location and Frequency of Sampling

Methyl Bay was sampled eight times, at about weekly intervals, between July 4 and August 25. Cedar and Granville lakes were each sampled four times, at about 2 week intervals, between July 9 and August 21. On each survey, four stations were sampled in each of Methyl Bay and Granville Lake and six stations were sampled in Cedar lake. Stations were classified as inshore or offshore and as flooded or unflooded (Table 1). Flooded stations were located over inundated terrestrial material in the reservoirs as determined from depth measurements and the estimated depth of flooding (Brown 1974, Water Resources Branch 1974, Green and Derkson 1980, Newbury et al. 1984). In the reservoirs, unflooded stations were located over fine-grained sediments within the pre-flooding lake basin. Inshore stations were usually within 5 m of the lake shore. Offshore stations were generally 300-400 metres directly offshore from the inshore stations at each site.

Samples Collected

Mercury methylation and methyl mercury demethylation rates were measured on samples of surficial lake bottom sediment from all stations on each survey. The samples (2L volume) were collected using a 12 volt submersible bilge pump

(Attwood model 4103, 360 gallons per hour) attached to a 15 m length of 2 cm (ID) Tygon tubing.

Plankton samples were collected from each lake on all surveys for analysis of taxonomic composition and total mercury concentration. The samples for mercury analysis were collected by towing Wisconsin-style nets (25 cm mouth diameter, 1 m in length) horizontally for 5-10 minutes about 20 m behind the float plane or boat between the inshore and offshore stations. Duplicate samples were collected in this manner on all surveys using nets constructed with each of 53, 150, and 300 um mesh-size Nitex. The samples collected in each net were bottled separately and kept on ice for return to the field lab. Later the same day, the volume of each sample was reduced by seiving through a 50 um mesh-size Nitex screen and the sample was dried to constant weight at 60° C .

Profiles of temperature, conductivity, and dissolved oxygen concentration were recorded at each station using a Whitney model CTU-3B thermistor -conductivity meter and a YSI model 54A thermistor-oxygen meter. Water transparency was measured at each station with a 20 cm diameter black and white Secchi disc.

Laboratory Analyses

Sample handling and incubation

Sediment samples were analysed for particulate carbon (as total carbon, organic carbon, and carbonate carbon), nitrogen (as total nitrogen), and total mercury concentrations and for pH. Oxygen consumption and/or production was measured during the sample incubation period.

In the laboratory, the sediment was mixed and measured into 250 mL BOD bottles for oxygen measurements and 125 mL glass reagent bottles for the measurement of specific mercury methylation and demethylation. Oxygen was measured at the beginning and end of the incubation period using the modified Winkler method (Stainton et al. 1977). Each of the specific methylation and demethylation measurements were determined using three replicate subsamples (one blank and a pair of replicates). The microbial activity in the blanks was stopped with 3 mL of 4N HCl prior to radioisotope addition. Methylation samples were injected with ^{203}Hg as $HgCl_2$ at a dose of 74kBq 2ug Hg^{-1} g dry $sediment^{-1}$. Demethylation samples were injected with ^{14}C as CH_3HgI at a dose of 1.55kBq 0.96 ug Hg^{-1} g dry $sediment^{-1}$. The samples were shaken for one minute after isotope addition then incubated in the dark at close to in situ temperatures (Table 1) for 12-16 hours. Samples were incubated with air in the head space of the incubation vessels and were not shaken or stirred during the incubation period.

Specific rate of mercury methylation

The specific rate of methylation was determined using the method of Furutani and Rudd (1980). After the samples were incubated and the microbial activity halted with 4 N HCl, the alkylated ^{203}Hg was extracted from the sediment using an organic extraction method (Furutani and Rudd 1980).

Specific rate of methyl mercury demethylation

The specific rate of demethylation was measured using the method of Ramlal et al. 1986a. At the end of the incubation period, the volatile ^{14}C produced by demethylation was stripped from acidified samples, oxidized to $^{14}CO_2$, and trapped in a carbon dioxide trap consisting of the fluor PCS (Amersham), methanol, and CO_2mMet, a carbon dioxide trapping agent (Amersham) (Ramlal et al. 1986a).

Calculations

The rate of methylation in sediment was first calculated in ng Hg methylated g^{-1} h^{-1}. This value was then used to calculate the specific rate of methylation, i.e. the percent of the added mercury methylated g^{-1} h^{-1}. The specific rate of demethylation was calculated in a similar manner (Ramlal et al. 1986a). The ratio of specific methylation to specific demethylation (M/D) was calculated by comparing the rates for the two processes:

$$M/D = \frac{\text{\% added Hg methylated } g^{-1}\ h^{-1}}{\text{\% added Hg demethylated } g^{-1}\ h^{-1}}$$

Plankton mercury concentrations

The concentration of total mercury in the dried plankton samples was determined on duplicate subsamples using the method of Armstrong and Uthe (1971). This method proved inaccurate for sample weights below 20 mg (dry wt.). The weights of samples from Cedar Lake collected in the 300 um mesh-size net occasionally were below this limit and in these cases the results of the analyses were rejected.

Results

Physical measurements

In Methyl Bay, maximum water temperatures (surface, mean water column, and bottom) were recorded at the inshore stations in the first week of July. Minimum water temperature occurred in the third week of July (Appendix Table 1, Fig. 3). At the offshore station, the highest surface and mean water column temperatures were recorded in the first week of July as well. Bottom temperatures peaked in mid July and varied little for the remainder of the sampling period (Appendix Table 1, Fig. 3). In general, temperatures were within 1° C of the seasonal mean throughout the period of study at all stations.

Water temperatures (surface, mean water column, bottom) in Cedar Lake peaked in the first week of August at all stations. Bottom temperatures were generally within 1° C of surface temperatures (Appendix Table 1, Fig. 4). The maximum temperature fluctuation over the period of study was about 4° C.

Maximum water temperatures (surface, mean water column, and bottom) were also recorded in Granville lake during the first week of August; peaking about 2° C higher than the minimum temperatures recorded at the start of sampling (Appendix Table 1, Fig. 5). Bottom temperatures were generally within 1° C of surface temperatures at all stations on all sampling dates.

Cedar Lake was the most turbid at the start of sampling, with a Secchi disc visibility of 0.6-0.7 m (Fig. 6). Turbidity decreased over the course of the season, with Secchi disc visibilites exceeding 1.4 m at some stations at the end of August. In Granville Lake, Secchi disc visibility fluctuated about the 1 m level for the sampling period (Fig. 6). Water transparency in Methyl Bay was comparable to that in Granville Lake at the start of sampling but declined through July, reaching about 0.6 m in early August and remaining near this level to the end of the month (Fig. 6). This increase in turbidity occurred as water levels rose to near-record highs (G. McCullough, Department of Fisheries and Oceans, Freshwater Institute, Winnipeg, Manitoba pers. com.) in July.

The water columns of all three sites were generally well-oxygenated throughout the study. The near bottom waters in both Granville and Cedar lakes were never seriously depleted of oxygen. Low ((2 ppm) near bottom oxygen levels were occasionally recorded in Methyl Bay but these never persisted (Appendix Table 1).

Water column conductivity was highest in Cedar Lake and lowest in Granville Lake in early July (Appendix Table 1). A malfunction of the instrument in early August, indicated by the convergence of values to about 100 uS cm^{-1}, prevented collection of a record for the entire sampling period. However, past studies have shown that conductivity varies little over the season (Ramsey and Ramlal 1985) so these few measurements should adequately characterize differences in conductivity among the lakes.

Sediment chemistry

The highest sediment total mercury concentrations were recorded during the first half of July in the three lakes (Tables 2, 3, and 4) particularly at the inshore stations in Methyl Bay. Following these high initial levels, total mercury concentrations in Granville Lake sediments averaged 35-55 ng g^{-1} both inshore and offshore (Table 2). Similarly, total mercury levels in sediments from the offshore station in Methyl Bay and from the inshore stations in Cedar Lake averaged 40-50 ng g^{-1} after mid-July (Table 3). Mercury levels in sediments from the flooded stations in Methyl Bay were notably higher, averaging about 95 ng g^{-1} after the middle of July (Table 3), as were levels at the offshore stations in Cedar Lake, which averaged about 100 ng g^{-1} (Table 4).

The total carbon content of surficial sediments from Granville Lake was similar at the inshore and offshore stations and was generally in the 30-40 mg g^{-1} range (Table 5). Total carbon concentrations in offshore Methyl Bay sediments averaged about 65% of the Granville Lake values (Table 5). In contrast, concentrations in sediments from the flooded stations in Methyl Bay were about 6 times those offshore and about 4 times the Granville Lake values

(Table 5). With similar carbonate concentrations at all stations in Methyl Bay and Granville Lake, the variation in total carbon content among the sites was due to differences in the size of the organic fraction (Table 5). Total carbon concentrations in Cedar lake sediments were generally similar to those in Granville Lake with no consistent difference between the inshore and offshore stations (Table 5). However, carbonate concentrations were 4-5 times higher so organic carbon levels were about 70% of those in Granville Lake (Table 5).

The sediment total nitrogen concentrations followed the rank order of the organic carbon concentrations among the lakes with the highest levels at the flooded Methyl Bay stations and the lowest in Cedar Lake and at the offshore station in Methyl Bay (Table 5).

Methyl Bay and Granville Lake sediments were of comparable pH. Values were in the circumneutral range throughout the study and there was no consistent difference between the inshore and offshore stations in either lake (Tables 6 and 7). The pH of Cedar Lake sediments was always higher than in the other lakes, usually by about 1 unit. As in the other lakes, there was never a marked difference in pH between the inshore and offshore stations in Cedar Lake (Tables 6, 7, and 8).

Biological activity of sediments

On most sampling dates, and particularly during the period of maximum specific methylation, the biological oxygen demand of sediments from the inshore stations in Methyl Bay was substantially higher than in offshore sediments (Table 6). The oxygen demand of sediments from Granville Lake was comparable to that of inshore sediments from Methyl Bay but there was no comparable trend of lower activity in offshore Granville Lake sediments (Tables 6 and 7). Biological oxygen demand in sediments from Cedar Lake was lower than in Granville Lake or in inshore sediments from Methyl Bay (Tables 6, 7, and 8).

Concentration of mercury in plankton

On average, there was no difference in the total mercury concentration among plankton samples collected in the different mesh-sized nets in any of the lakes (Fig. 7). Furthermore, there was no difference, on average, in the total mercury concentration among plankton samples collected from the three sites, with concentrations averaging between 0.05 and 0.06 ug g^{-1} (dry wt.) (Fig. 7). There was little fluctuation of plankton mercury levels over the season in any of the lakes (Fig. 8). All of the collected data are tabulated in Appendix Table 4.

Specific rate measurements

Granville Lake

At the inshore stations, rates of specific methylation were low but detectable on all sampling dates with the highest rates measured at the end of July (Table 9, Fig. 9). Methylation was detectable at the offshore station only on July 9. Specific demethylation followed a distinct pattern of seasonal fluctuation with peak rates recorded in about mid-August (Fig.10). The maximum rate of demethylation measured at the offshore station was about twice the peak rate recorded at the inshore stations and this differential was reflected in the seasonal mean values (Table 9). Inshore, M/D varied inversely with demethylation and the peak value was recorded in late August (Fig.11). As a result of the negligible methylation rates, M/D at the offshore station was at or near zero for the entire sampling period (Fig.11).

Methyl Bay

There was considerable variation in methylation and demethylation rates among the stations on every sampling date but the rates of specific methylation and demethylation were always greater in the flooded zone (Table 10). At the inshore stations both methylation and demethylation followed distinct seasonal patterns of fluctuation (Fig.9 & 10.Specific methylation rates peaked at the end of July while specific demethylation peaked in mid to late August. Although the data indicate the peak M/D ratio occurred in early July this was probably the result of the missing demethylation data for the late July survey and the peak more likely coincided with peak methylation. At the offshore station, methylation was detectable only on July 29 when the peak rates were measured inshore. Demethylation also was maximal in mid to late August at the offshore station, although the rate was just 15% of that recorded inshore.

On average, inshore methylation rates were more than 4 times higher than in Granville Lake (Tables 9 and 10). Specific demethylation in the flooded zone averaged almost 3 times the inshore rates in Granville Lake. However, due to more frequent sampling during the peak of demethylation in Methyl Bay than in Granville Lake (Fig. 10), this is likely an overestimate. Inshore M/D in Methyl Bay averaged 1.4 times that in Granville Lake (Tables 9 and 10).

The specific methylation rate measured in offshore Methyl Bay sediments was, on average, comparable to that in Granville Lake (Tables 9 and 10). In contrast, specific demethylation at the offshore station was about 20% of the rate in Granville Lake. Again due to the generally unmeasurable methylation rates, M/D was at or near zero for the entire sampling period (Fig. 11).

Cedar Lake

Although all stations were located over flooded material, specific methylation rates were generally very low, exceeding those in Granville Lake only in late August (Tables 9 and 11, Fig. 9). This was the only time substantial methylation was detected and then only in two of the eight samples. The maximum methylation rate recorded was less than half the peak rate measured in Methyl Bay (Tables 10 and 11). Inshore rates were not consistently different from those measured at offshore stations. Specific demethylation rates varied considerably among the stations. Demethylation tended to be highest in early August but the seasonal trend was much less distinct than in Granville Lake or Methyl Bay (Fig. 10). Consequently, the peak M/D coincided with peak methylation rates (Fig. 11).

Discussion

Mercury concentrations in plankton

The concentration of mercury in plankton was used as a relative measure of the water total mercury concentration in the absence of a method with sufficient sensitivity to measure the concentration in the water directly. As found in the 1984 survey (Appendix 3), the concentrations of total mercury in plankton indicate that reservoir formation has had no effect on concentrations of total mercury in the water column nor are concentrations particularly high (Fig. 7). Again this year, there was no difference in the total mercury concentrations among plankton samples collected with 53, 150, and 300 um mesh-size nets at any of the sites. This is in keeping with the results of other studies (Sarkka et al. 1978; Skei et al. 1976; Knauer and Martin 1972; Cocoros et al. 1973) and confirms that the total mercury concentrations found in the plankton samples are proportional to the concentration in the water column.

Unlike the 1984 survey, there was no distinct seasonal fluctuation in the plankton mercury concentrations at any of the sites (Fig. 8). The large pulse of mercury which appeared in the plankton samples in the second half of August in 1984 was hypothesized to be the result of a flux from the sediments into the overlying waters (Appendix 3). The cause of this flux was suggested to be fall cooling which can promote thermal instability in the upper interstitial water column causing circulation within the sediments and a flushing of dissolved constiutents (Hecky et al. 1986). There was no distinct cooling period at any of the sites during our period of observation in 1985 (Fig. 3, 4, and 5). As a result there was no opportunity for thermal instability to develop in the sediment interstitial waters and, with no circulation, no mercury would be flushed into the overlying water column. The lack of a mercury pulse in 1985 also appears to account for the lower mean plankton mercury concentrations in Methyl Bay and Granville Lake than were observed in 1984 (Appendix 3).

Methylating activity in Cedar Lake

Cedar Lake does not presently support elevated rates of biological net mercury methylation as indicated by specific methylation rates and M/D ratios which are comparable to those in Granville Lake (Tables 9 and 11). The low methylating activity is consistent with the low fish mercury levels currently found in the lake (Fig. 1) and is consistent with the decrease in fish mercury concentrations in the mid 1970's. This finding also supports our hypothesis that fish mercury levels in lakes flooded by the Churchill River Diversion will not return to pre-impoundment levels until the presently high rates of methylating activity in the lakes decline (Appendix 3).

Although the findings of this study are in keeping with the current state of fish mercury levels in Cedar Lake, the question of why the problem was so short-lived (about 12 years) in comparison to our projection for the duration of the problem in SIL and Notigi (Appendix 3) must be reconciled. Based on the results of the 1984 survey of reservoirs in northern Manitoba, we speculated that it was the flooded organic material which was stimulating methylating activity and that the mercury problem would persist until this material was consumed by decomposition or flushed from the lakes, with the former pathway the more important. Decomposition will proceed slowly in lakes flooded by the diversion because trees accounted for over 90% of the flooded vegetation biomass.

A very different vegetation zone was flooded by the impoundment of Cedar Lake. The majority (est. >80%) of land flooded was in the area of the Saskatchewan River Delta (Green and Derksen 1980). Although the vegetation cover in this region has not been mapped extensively, photographs of the surrounding area (Fig. 12) and the Environmental Impact Assessment for the project (Anonymous 1961) indicate that, with the exception of the river and stream levees, it was almost solely marsh. The organic loading per unit flooded area from this type of vegetation cover should have been considerably lower than from the productive forest which predominated the nutrient inputs to lakes flooded by the diversion (G. McCullough, pers. com.; R. Uchtmann, Marsh Manager; The Pas Region, Manitoba Department of Natural Resources, The Pas, Manitoba, pers. com.) although, in the absence of adequate vegetation maps, it is difficult to quantify this difference. However, the magnitude of the difference in organic loading might be approximated by asssuming the nutrient content of the vegetation flooded in Cedar Lake was comparable to that of treed muskeg in the SIL area. (G. McCullough, pers. com.) In this case the nutrient contribution to Cedar Lake per unit of flooded area would have been

10% of that in SIL or Notigi Reservoir. This approximation is most likely an underestimate of the actual difference in nutrient loading due to flooding as the standing biomass of flooded muskeg probably overestimates that of the macrophytes and sedges in the flooded marshland (G. McCullough, R. Uchtmann pers. com.).

It should also be expected that the types of vegetation flooded in Cedar Lake would degrade quickly relative to the material introduced to the Churchill River diversion lakes (Hodkinson 1975; Anderson et al. 1978) although this difference cannot be quantified with available information. , Flooded trees were still evident in Cedar Lake (Fig. 12) but the low sediment organic carbon concentrations (Table 5) compared to Methyl Bay indicate that other flooded vegetation has decomposed. This was also apparent from examination of sediment samples collected with an Ekman dredge. These consisted of sand, a small amount of organic sediment, and roots of submerged macrophytes which were prevalent at the sampling site. None of the material had an obvious terrestrial origin. This is quite in contrast to Reindeer Lake where flooded trees and wood chips still litter the nearshore waters and the sediment organic carbon concentrations are comparable to those in the flooded zone of SIL 42 years after impoundment (Appendix 3). Together, the much lower loading of nutrients to Cedar Lake in the flooded vegetation and the apparently higher degradability of this material seem to be responsible for the relative transience of the fish mercury problem in Cedar Lake compared with that projected for lakes flooded by the Churchill River diversion.

Methylating activity in Southern Indian Lake

The flooded zone of Methyl Bay in Southern Indian Lake continues to support high specific rates of mercury methylation ten years after reservoir formation. In each of the past four years, methylation rates at flooded locations have substantially exceeded those at stations located in the preflooding lake basin (Fig. 13). Comparisons with methylating activity in Granville Lake in 1984 (Appendix 3) and now again in 1985 indicate that measurements made at the offshore station in Methyl Bay provide a good approximation of methylating activity in the sediments of Southern Indian Lake prior to flooding. There is no indication that the differential in specific methylation rates between the flooded and offshore stations in Methyl Bay is diminishing (Fig. 13). Year to year variation is of the same order as variation among the stations in a given year (Table 10; Appendix 3). The persistence of elevated methylation rates in Methyl Bay is consistent with the conclusion of our 1984 study that the fish mercury problem in Southern Indian Lake and Notigi Reservoir will be long-lived.

In contrast to methylation, specific demethylation rates can vary greatly from year to year. Seasonal mean demethylation rates were 2 to 6 times higher in Granville Lake and 2 to 11 times higher in Methyl Bay in 1985 than in 1984 (Table 12). While there was no difference between the rates measured at the inshore and offshore stations at either site in 1984, specific demethylation at the offshore station was twice the inshore value in Granville Lake but was less than 20% of the inshore value in Methyl Bay in 1985 (Table 12). Demethylation rates in the inshore zone of Methyl Bay were less than half those in Granville Lake in 1984 (Appendix 3) but were almost 3 times the Granville Lake rate in 1985 (Table 12). As a result, the seasonal mean M/D for the flooded stations in Methyl Bay was about 1.4 times the inshore M/D in Granville Lake (Tables 9 and 10), which is about 50% of the differential found in 1984 (Appendix 3).

The seasonal mean demethylation rates for the two lakes in 1985 may not be directly comparable due to the more frequent sampling of Methyl Bay than

Granville Lake during the period of peak demethylating activity (Fig. 10). Eliminating the measurements for which there is not a corresponding value in Granville Lake reduces the difference in mean demethylation to about a factor of 2. This revised estimate of the mean specific demethylation rate at the inshore stations in Methyl Bay makes the seasonal mean M/D about 1.9 times the inshore M/D in Granville Lake which is approximately the same differential as in 1984 (Appendix 3).

The reason for the generally higher rates of demethylation in 1985 is not apparent from any of the parameters we measured. These are potentially the result of higher rates of microbial activity but there is no reason for activity to have been higher in 1985 than in 1984. Water temperatures were generally lower in 1985 and sediment organic carbon concentrations have not increased since 1984.

The degradation of methylmercury is mediated by the enzyme organomercurial lyase (Robinson and Tuovinen 1984). Enzyme synthesis is triggered by the presence of a mercury-containing inducer substance. Therefore, it might be expected that specific demethylation rates would be higher at locations with higher sediment mercury concentrations. The highest rates were recorded at the location with the highest sediment total mercury concentration but differences in sediment mercury levels do not explain the variation in demethylation rates among the other sites nor do they explain the temporal variation observed at any of the sites (Tables 2-4; Fig. 10). Furthermore, surficial sediment mercury concentrations have not increased in the last year in either lake (Tables 2 and 3; Appendix 3).

Cause of the elevated specific methylation rates

In the past, increased supplies of inorganic mercury have been identified as the cause of fish mercury problems in reservoirs (Potter et al. 1975; Abernathy and Cumbie 1977; Meister et al. 1979; Cox et al. 1979) apparently on the assumption that methylation rates were limited by mercury availability. Several studies have demonstrated that additions of inorganic mercury can increase the rate of methylation (Rudd et al. 1983; Wright and Hamilton 1982). Methyl Bay had the highest sediment total mercury concentrations and the highest methylation rates which at least superficially supports the long-standing hypothesis of the cause. However, elevated sediment total mercury concentrations were ruled out as the cause of the higher methylations rates in 1984 because the rates were no higher than in the flooded zone of reservoirs where the sediment total mercury concentrations were the same as in unflooded reference lakes (Appendix 3). Furthermore, if increased mercury levels were the primary cause of the elevated methylating activity, rates of specific methylation at the offshore stations in Cedar Lake should have been higher than in Granville Lake but this was not the case (Tables 9 and 11, Fig. 9). The seasonal fluctuations in methylation rates also were not correlated with sediment total mercury concentrations.

Another possible cause of the high methylation rates is the introduction of organic material during reservoir formation. Numerous studies have shown that methylation rates are elevated by the addition of an organic substrate (Furutani and Rudd 1980; Wright and Hamilton 1982; Jensen and Jernelov 1969; Jernelov 1972). In the present study, high methylation rates occured only in locations with high sediment organic carbon concentrations (Tables 5 and 9-11). In Methyl Bay, the specific methylation rate and sediment organic carbon each averaged about 4 times the respective values for Granville Lake sediments. The only locations in Cedar Lake where methylation was measurable had organic carbon concentrations exceeding those of Granville lake sediments.

The addition of an organic substrate is thought to stimulate methylation by increasing microbial activity (Furutani and Rudd 1980; Wright and Hamilton 1982; Jensen and Jernelov 1969; Jernelov 1972). Consistent with this hypothesis, sediments from the inshore stations in Methyl Bay consumed oxygen at more than 4 times the rate measured in samples from the offshore station and at 1.5 to 11.5 times the rates of oxygen consumption in sediments from the other lakes (Tables 6, 7, and 8).

Although it seems clear that the influx of mercury to the reservoir in the flooded terrestrial soils and vegetation is not by itself responsible for the elevated rates of methylating activity, there is some evidence to suggest that it may magnify the effect of any increase in microbial activity due to the organic loading. In enclosure experiments conducted at SIL, Ramlal et al. (1986b) have found that additions of inorganic mercury as Hg^{2+} act to select for bacteria capable of methylating mercury. Coupled with an increase in general microbial activity, this selection would account for a greater increase in methylation rates than would be expected from the magnitude of the increase in bacterial growth rates alone.

Seasonal pattern of methylating activity

As in previous investigations, specific methylation rates followed a distinct seasonal pattern with maximum rates in mid- summer (Fig. 9). The cause of this seasonality is even less apparent from the measurements made this year than it was in 1984. We speculated that this pattern was the combined result of the seasonal fluctuation of water temperature and a flux of mercury from the sediments in the latter part of the summer (Appendix 3). However, there was neither a marked fluctuation in temperature nor a pulse of mercury from the sediments in 1985.

Conclusions

After four years of monitoring methylating activity in Metnyl Bay, there is no indication that the elevated rates of biological net methylation are declining within 10 years of impoundment. The measurements of methylation in Cedar Lake indicate that such a decline will be required before fish mercury levels begin to decrease. No evidence has appeared that would cause us to modify our projection (Appendix 3) that the methylation balance in Southern Indian Lake and Notigi Reservoir will not begin to return to the pre-flooding state for at least another 30 years.

Acknowledgements

We would like to thank Chris Hecky for his assistance in the field and with processing samples in the lab. The total mercury analyses were performed by R. Hunt and the carbon and nitrogen analyses were done by B. Hauser, both of the Freshwater Institute. Labelled methyl mercury was provided by J. Rudd.

This study was conducted under DSS contract file number 01SF.FP430-5-S9039 with funding provided by the Canada-Manitoba Agreement on the Study and Monitoring of Mercury in the Churchill River diversion and by the Department of Fisheries and Oceans.

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Table 1. Station types sampled on each survey and the incubation temperatures of the methylation and demethylation assays. I and O denote inshore and offshore locations, respectively. F and U denote flooded and unflooded locations.

Lake	Date	Inshore station Numbers	Inshore station Type	Offshore station Numbers	Offshore station Type	Incubation Temperature (°C)
Methyl Bay	July 4	1-3	flooded	4	unflooded	13
	July 15	1-3	(I-F)	4	(O-U)	15
	July 29	1-3		4		14
	August 6	1-3		4		15
	August 13	1-3		4		15
	August 19	1-3		4		15
	August 25	1-3		4		15
Cedar	July 10	2-5	flooded	1,6	flooded	18
	July 24	3-5	(I-F)	1-2	(O-F)	17
	August 7	3-6		1-2		19
	August 21	1-6		-		16
Granville	July 9	1-3	unflooded	4	unflooded	16
	July 30	1-3	(I-U)	4	(O-U)	16
	August 7	1-3		4		18
	August 21	1-3		4		16

Table 2. Total mercury concentrations in Granville Lake sediments. Values labelled A and B are results of analyses of duplicate subsamples.

		Range			Mean	
Date	Station	A	B	Average	Inshore	Offshore
			ng Hg (g dry sediment)$^{-1}$			
July 9	1	168	259	213	202	80
	2	107	113	110		
	3	NA	283	283		
	4	80	NA	80		
July 30	1	21	32	27	33	40
	2	36	49	43		
	3	23	35	29		
	4	36	43	40		
August 8	1	55	31	43	45	37
	2	70	42	56		
	3	36	36	36		
	4	37	NA	37		
August 20	1	29	30	29	28	50
	2	32	35	33		
	3	30	15	23		
	4	44	57	50		

Table 3. Total mercury concentrations in Methyl Bay sediment samples. Values labelled A and B are results of analyses of duplicate subsamples.

		Range			Mean	
Date	Station	A	B	Average	Inshore	Offshore
			ng Hg (g dry sediment)$^{-1}$			
July 4	1	143	223	183	203	56
	2	213	283	248		
	3	450	164	307		
	4	66	45	56		
July 15	1	169	NA	169	599	589
	2	987	437	712		
	3	609	383	496		
	4	934	244	589		
July 22	1	51	93	72	61	47
	2	76	95	86		
	3	44	56	50		
	4	48	45	47		
July 29	1	71	151	111	119	44
	2	57	144	101		
	3	144	NA	144		
	4	43	44	44		
August 6	1	160	94	127	91	50
	2	99	123	111		
	3	127	99	113		
	4	49	51	50		
August 13	1	78	NA	78	99	36
	2	151	155	153		
	3	64	70	67		
	4	37	35	36		
August 19	1	98	97	97	76	45
	2	87	103	95		
	3	87	88	88		
	4	51	38	45		
August 25	1	71	138	104	134	38
	2	92	212	152		
	3	130	291	211		
	4	42	33	38		

Table 4. Total mercury concentrations in Cedar Lake sediments. Values A and B are results of analyses of duplicate subsamples.

		Range			Mean	
Date	Station	A	B	Average	Inshore	Offshore
			ng Hg (g dry sediment)$^{-1}$			
July 10	1	80	NA	80	87	118
	2	26	26	26		
	3	28	NA	28		
	4	100	99	100		
	5	82	72	77		
	5	264	143	203		
	6	NA	43	43		
	6	181	282	231		
July 24	1	84	151	121	30	73
	2	24	NA	24		
	3	13	26	19		
	4	12	28	20		
	4	23	50	37		
	5	31	59	45		
	5	26	51	38		
August 7	1	124	315	220	41	116
	2	65	54	60		
	2	66	74	70		
	3	42	30	36		
	4	22	43	32		
	4	30	40	35		
	5	24	62	43		
	6	52	61	57		
August 21	1	20	56	38	64	
	2	98	158	128		
	2	27	47	37		
	3	17	47	32		
	4	49	81	65		
	4	74	92	84		
	5	77	98	88		
	6	38	41	40		

Table 5. Comparison of carbon and nitrogen contents of sediments from Methyl Bay, Cedar Lake, and Granville Lake. Means calculated from data in Appendix 3. All values expressed as mg g sed dw^{-1}.

Lake	Date	Station Type	Mean Total Carbon	Mean Carbonate	Mean Organic Carbon	Mean Total Nitrogen
Methyl Bay	July 4	I-F	93	3	89	6.0
		O-U	22	1	21	3.6
	15	I-F	86	2	84	5.2
		O-U	23	1	22	3.3
	22	I-F	92	4	88	5.0
		O-U	20	1	19	2.7
	29	I-F	146	4	142	6.7
		O-U	20	1	19	2.7
	August 6	I-F	146	4	142	7.0
		O-U	22	1	21	3.1
	13	I-F	120	4	116	5.5
		O-U	19	1	18	2.0
	19	I-F	130	2	128	6.3
		O-U	22	1	21	2.8
	25	I-F	150	3	147	6.9
		O-U	23	1	22	3.2
Cedar	July 10	I-F	37	10	27	1.8
		O-F	31	7	24	1.8
	24	I-F	25	13	12	0.8
		O-F	60	13	47	3.9
	August 7	I-F	25	16	9	1.0
		O-F	66	13	53	5.2
	21	I-F	95	13	82	5.6
Granville	July 9	I-U	63	2	61	6.1
		O-U	37	1	36	5.3
	30	I-U	28	1	27	3.5
		O-U	21	1	20	2.8
	August 8	I-U	33	2	31	4.5
		O-U	32	2	30	4.7
	20	I-U	32	1	31	4.3
		O-U	36	1	35	4.9

Table 6. Measurements of pH and oxygen demand of sediment samples from Granville Lake. Negative oxygen values indicate a net production of oxygen during the incubation period. Station types as defined in Table 1.

Date	Station	Type	mg O_2 g^{-1} d^{-1}	Average	pH
July 9	1	I-U	NA	0.542	6.86
	2	I-U	0.713		6.98
	3	I-U	0.372		7.18
	4	O-U	0.701	0.701	7.12
August 8	1	I-U	0.008	0.165	6.76
	2	I-U	0.065		6.82
	3	I-U	0.421		6.98
	4	O-U	0.676	0.676	7.14
August 20	1	I-U	0.081	0.134	6.72
	2	I-U	0.077		7.02
	3	I-U	0.245		7.10
	4	O-U	-0.355	-0.355	7.18

Table 7. Measurements of pH and oxygen demand of sediment samples from Methyl Bay . Negative oxygen demand values represent net production of oxygen during the incubation period. Station types as defined in Table 1.

Date	Station	Type	mg O_2 g^{-1} d^{-1}	Average	pH
July 4	1	I-F	0.202	0.374	6.89
	2	I-F	0.520		7.18
	3	I-F	0.399		6.96
	4	O-U	-0.014	-0.014	7.02
July 15	1	I-F	0.023	0.163	6.52
	2	I-F	0.029		6.54
	3	I-F	0.436		7.42
	4	O-U	0.188	0.188	7.28
July 22	1	I-F	0.298	0.154	6.86
	2	I-F	0.164		6.78
	3	I-F	0		7.18
	4	O-U	0.247	0.247	7.22
July 29	1	I-F	0.354	0.910	7.08
	2	I-F	0.481		6.84
	3	I-F	1.895		7.02
	4	O-U	-0.024	-0.024	7.14
August 6	1	I-F	0.453	0.374	6.78
	2	I-F	0.290		6.88
	3	I-F	0.378		6.78
	4	O-U	0.111	0.111	7.66
August 13	1	I-F	0.099	0.285	7.00
	2	I-F	0.474		6.90
	3	I-F	0.281		7.08
	4	O-U	0.060	0.060	7.08
August 19	1	I-F	0.193	0.116	6.99
	2	I-F	0.116		6.82
	3	I-F	0.039		7.12
	4	O-U	0.098	0.098	7.02
August 25	1	I-F	0.557	1.312	6.99
	2	I-F	0.807		6.92
	3	I-F	2.571		6.94
	4	O-U	0.184	0.184	7.10

Table 8. Measurements of pH and oxygen demand of sediment samples from Cedar Lake. Negative oxygen values indicate net production of oxygen during the incubation period. Station types as defined in Table 1.

Date	Station	Type	mg O_2 g^{-1} d^{-1}	Average	pH
July 10	1	O-F	0.684	0.382	8.12
	2	I-F	-0.076	0.062	7.88
	3	I-F	-0.039		7.92
	4	I-F	0.052		7.96
	5	I-F	0.187		7.94
	6	O-F	0.232		8.22
July 24	1	O-F	0.250	0.268	8.20
	2	O-F	0.286		8.28
	3	I-F	0.048	0.199	8.28
	4	I-F	0.618		7.92
	5	I-F	0.142		7.84
August 21	1	I-F	-0.381	-0.142	7.42
	2	I-F	-0.402		8.12
	3	I-F	-0.184		8.21
	4	I-F	0.093		7.99
	5	I-F	0		7.58
	6	I-F	0.047		7.52

Table 9. Results of methylation (M) and demethylation (D) rate assays on sediment samples from Granville Lake. M/D is the ratio of specific methylation to specific demethylation. Station types as defined in Table 1.

	Station		% added mercury g^{-1} h^{-1}					
Date	Type	No.	M	Mean M	D	Mean D	M/D	Mean M/D
July 9	I-U	1	0.001	0.001	1.903	0.916	0.001	0.001
	I-U	2	0.002		0.561		0.004	
	I-U	3	0		0.285		0	
	O-U	4	0.002	0.002	0.610	0.610	0.003	0.003
July 30	I-U	1	0.001	0.009	0.652	0.346	0.002	0.046
	I-U	2	0.018		0.217		0.083	
	I-U	3	0.009		0.168		0.054	
	O-U	4	0	0	0.336	0.336	0	0
August 9	I-U	1	0.004	0.008	0.789	1.206	0.005	0.007
	I-U	2	0.018		1.322		0.014	
	I-U	3	0.002		1.506		0.001	
	O-U	4	0	0	2.236	2.236	0	0
August 20	I-U	1	0.011	0.004	0.026	0.077	0.423	0.145
	I-U	2	0		0.044		0	
	I-U	3	0.002		0.162		0.012	
	O-U	4	0	0	0.234	0.234	0	0
Seasonal Mean		1-3	0.006		0.375		0.016	
(Time-weighted)		4	<0.001		0.865		0.006	

Table 10. Results of methylation (M) and demethylation (D) rate assays on sediment samples from Methyl Bay. M/D is the ratio of specific methylation to specific demethylation. Station types as defined in Table 1.

	Station		% added mercury g^{-1} h^{-1}					
Date	Type	No.	M	Mean M	D	Mean D	M/D	Mean M/D
July 4	I-F	1	0.016	0.017	0.267	0.210	0.060	0.081
	I-F	2	0.006		0.113		0.053	
	I-F	3	0.029		0.251		0.116	
	O-U	4	0	0	0.016	0.016	0	0
July 15	I-F	1	0.020	0.031	0.224	0.289	0.089	0.107
	I-F	2	0.072		0.258		0.279	
	I-F	3	0		0.386		0	
	O-U	4	0	0	0.108	0.108	0	0
July 29	I-F	3	0.047					
	O-U	4	0.001					
August 6	I-F	1	0.002	0.030	1.795	1.243	0.001	0.024
	I-F	2	0.012		0.943		0.013	
	I-F	3	0.076		0.992		0.077	
	O-U	4	0	0	0.190	0.190	0	0
August 13	I-F	1	0.002	0.013	1.460	2.414	0.001	0.005
	I-F	2	0.002		3.304		0.001	
	I-F	3	0.034		2.477		0.014	
	O-U	4	0	0	0.397	0.397	0	0
August 19	I-F	1	0	<0.001	3.496	2.560	0	0
	I-F	2	0.001		1.436		0.001	
	I-F	3	0		2.748		0	
	O-U	4	0	0	0.358	0.358	0	0
August 25	I-F	1	0.001	0.010	0.849	0.679	0.001	0.017
	I-F	2	0.002		0.570		0.004	
	I-F	3	0.028		0.619		0.045	
	O-U	4	0	0	0.042	0.042	0	0
Seasonal Mean		1-3	0.026		1.097		0.023	
(Time-weighted)		4	<0.001		0.182		0.001	

Table 11. Results of methylation (M) and demethylation (D) rate assays on sediment samples from Cedar Lake. M/D is the ratio of specific methylation to specific demethylation. Station types as defined in Table 1.

	Station		% added mercury g^{-1} h^{-1}					
Date	Type	No.	M	Mean M	D	Mean D	M/D	Mean M/D
July 10	O-F	1	0	0.001	0.937	0.508	0	0.002
	I-F	2	0	0	0.126	0.432	0	0
	I-F	3	0		0.419		0	
	I-F	4	0		0.462		0	
	I-F	5	0		0.209		0	
	I-F	5	0.001		0.946		0.001	
	O-F	6	0.001		0.296		0.003	
	O-F	6	0.001		0.292		0.003	
July 24	O-F	1	0	0				
	O-F	2	0					
	I-F	3	0	0.002				
	I-F	3	0					
	I-F	4	0					
	I-F	5	0.006					
August 7	O-F	1	0	0.006	2.504	0.967	0	0.035
	O-F	2	0.006		0.247		0.024	
	O-F	2	0.012		0.151		0.079	
	I-F	3	0	0	1.074	0.942	0	0
	I-F	4	0		0.496		0	
	I-F	4	0		0.603		0	
	I-F	5	0		1.695		0	
	I-F	6	0		0.841		0	
August 21	I-F	1	0	0.015	0.275	0.307	0	0.098
	I-F	2	0.002		0.852		0.002	
	I-F	2	0		0.095		0	
	I-F	3	0		0.343		0	
	I-F	4	0.001		0.501		0.002	
	I-F	4	0.038		0.145		0.262	
	I-F	5	0.076		0.180		0.422	
	I-F	6	0.006		0.061		0.098	
Seasonal Mean		I-F	0.003		0.666		0.005	
(Time-weighted)		O-F	0.002		0.738		0.002	

Table 12. Comparison of seasonal mean specific demethylation rates (% added Hg g^{-1} h^{-1}) measured in Granville Lake and Methyl Bay in 1984 and 1985 (based on all measurements). Data for 1984 from Appendix 3.

	1984		1985	
	inshore	offshore	inshore	offshore
Granville L.	0.185	0.132	0.375	0.865
Methyl Bay	0.096	0.086	1.097	0.182

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4. Seasonal fluctuations of mean water column and near-bottom temperature in Cedar Lake.

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12. Photographs of the sampling sites and shorelines in Cedar Lake and Methyl Bay.

13. Comparison of specific methylation rates measured at the flooded and unflooded stations in Methyl Bay; 1982-1984. Data from: Appendix 3, Ramlal et al. 1986b, and this study.

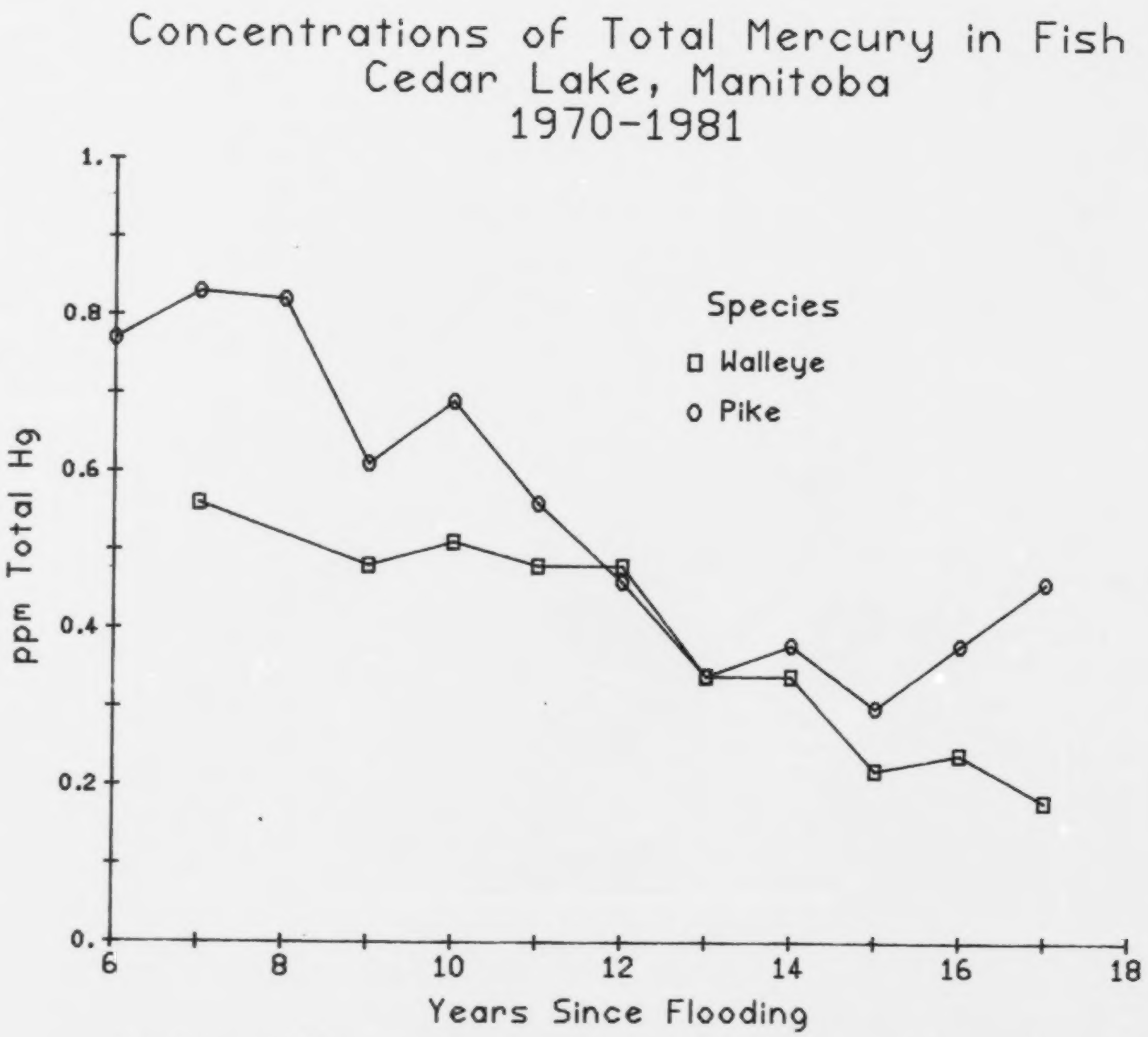
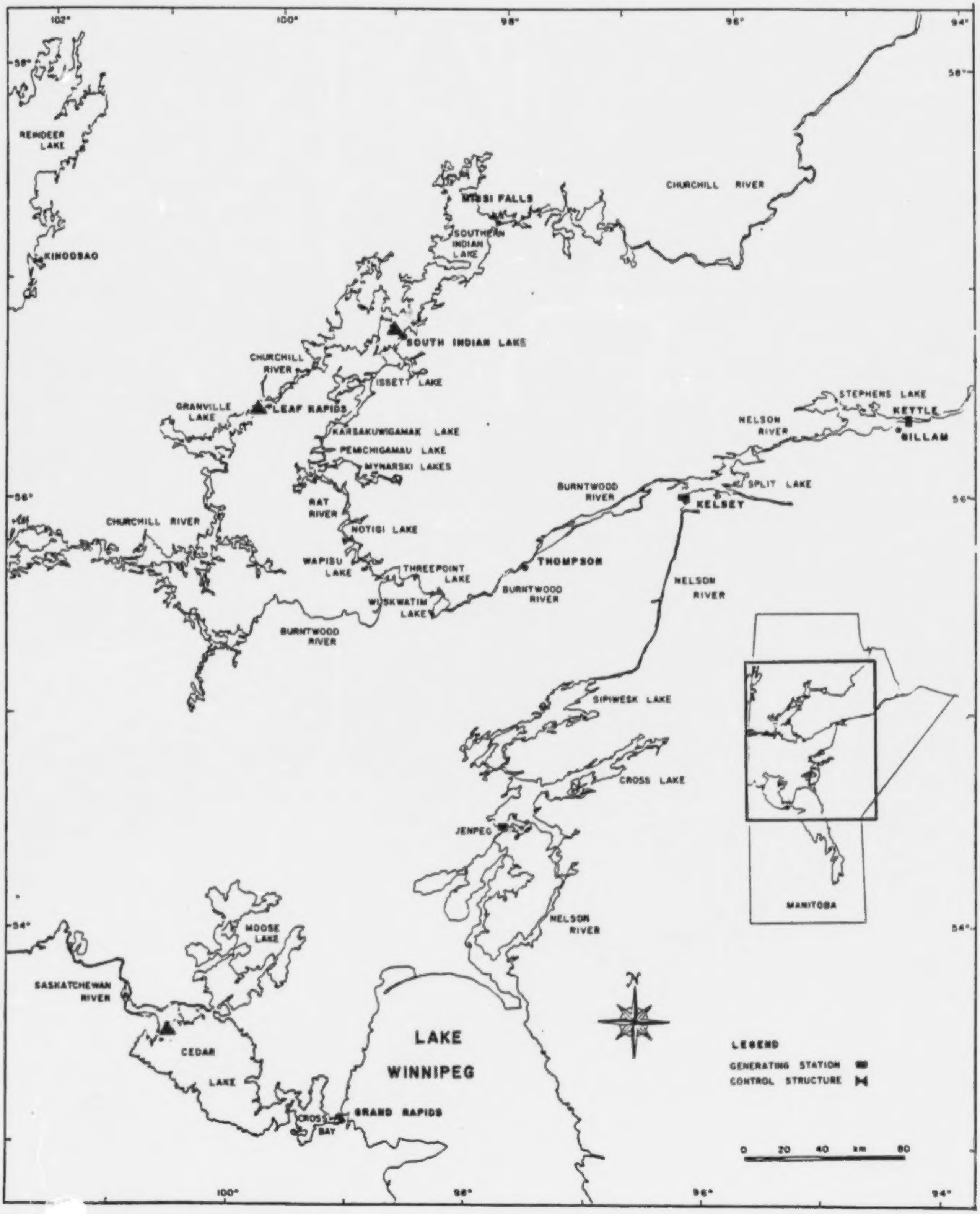


Fig. 1 Total mercury concentrations in Walleye and northern pike from Cedar Lake, Manitoba. Data from: Southern Operations Directorate, Department of Fisheries and Oceans, Freshwater Institute, Winnipeg, Manitoba.


102°
100°
98°
96°
94°
58°
56°
54°
REINDEER LAKE
KINOOSAO
MISSI FALLS
SOUTHERN INDIAN LAKE
CHURCHILL RIVER
SOUTH INDIAN LAKE
CHURCHILL RIVER
ISSETT LAKE
GRANVILLE LAKE
LEAF RAPIDS
KARSAKUWIGAMAK LAKE
PEMCHIGAMAU LAKE
MYNARSKI LAKES
RAT RIVER
NOTIGI LAKE
CHURCHILL RIVER
WAPISU LAKE
THREEPOINT LAKE
WUSKWATIM LAKE
BURNTWOOD RIVER
THOMPSON
BURNTWOOD RIVER
BURNTWOOD RIVER
STEPHENS LAKE
KETTLE
GILLAM
NELSON RIVER
SPLIT LAKE
KELSEY
NELSON RIVER
SIPIWESK LAKE
CROSS LAKE
JENPEG
NELSON RIVER
MANITOBA
MOOSE LAKE
SASKATCHEWAN RIVER
CEDAR LAKE
CROSS BAY
GRAND RAPIDS
LAKE WINNIPEG
N
LEGEND
GENERATING STATION
CONTROL STRUCTURE
0 20 40 km 80
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98°
96°
94°

Fig. 2 Location of sampling sites in Granville Lake, Cedar Lake, and Southern Indian Lake (Methyl Bay).

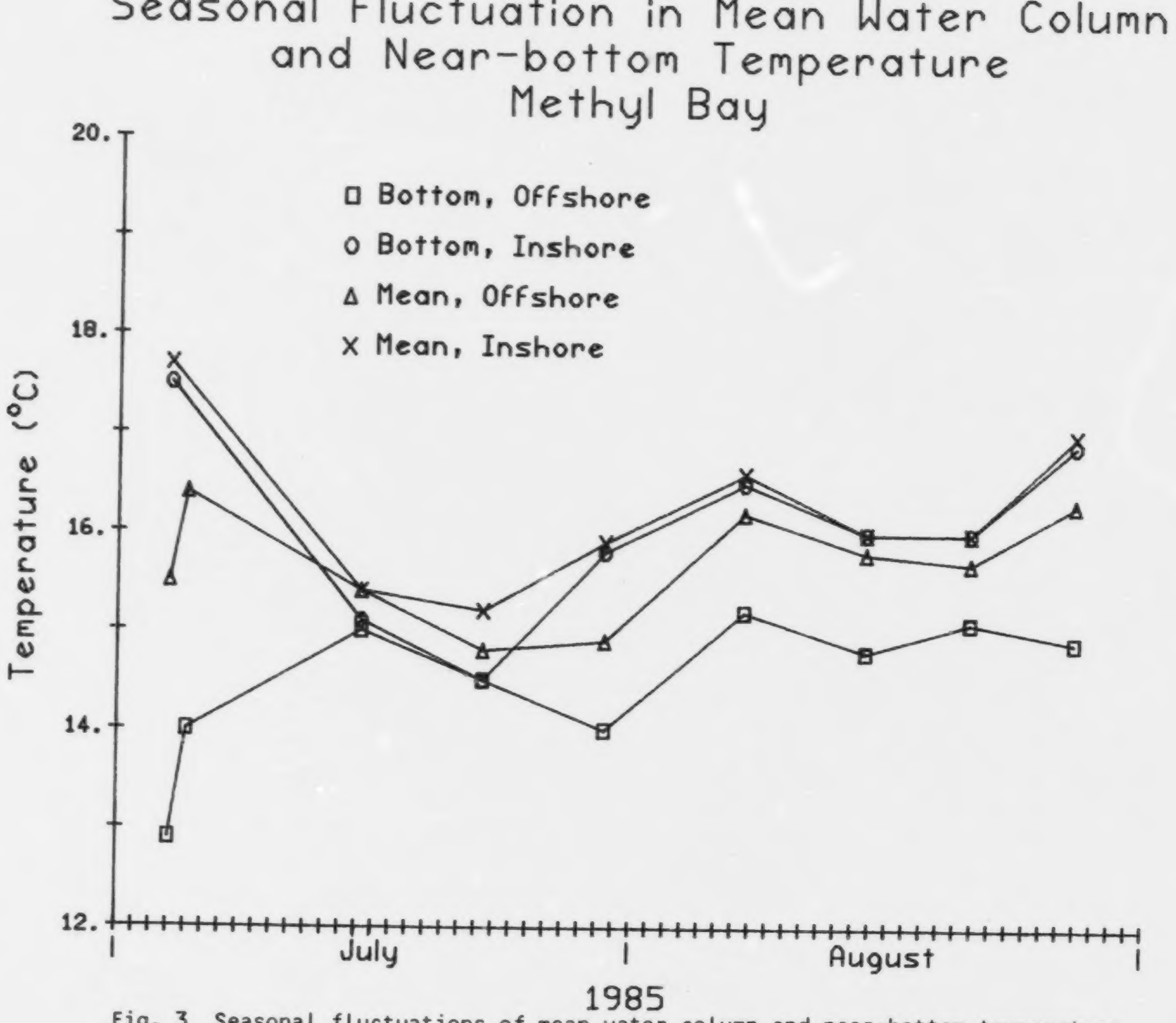


Fig. 3 Seasonal fluctuations of mean water column and near-bottom temperature in Methyl Bay.

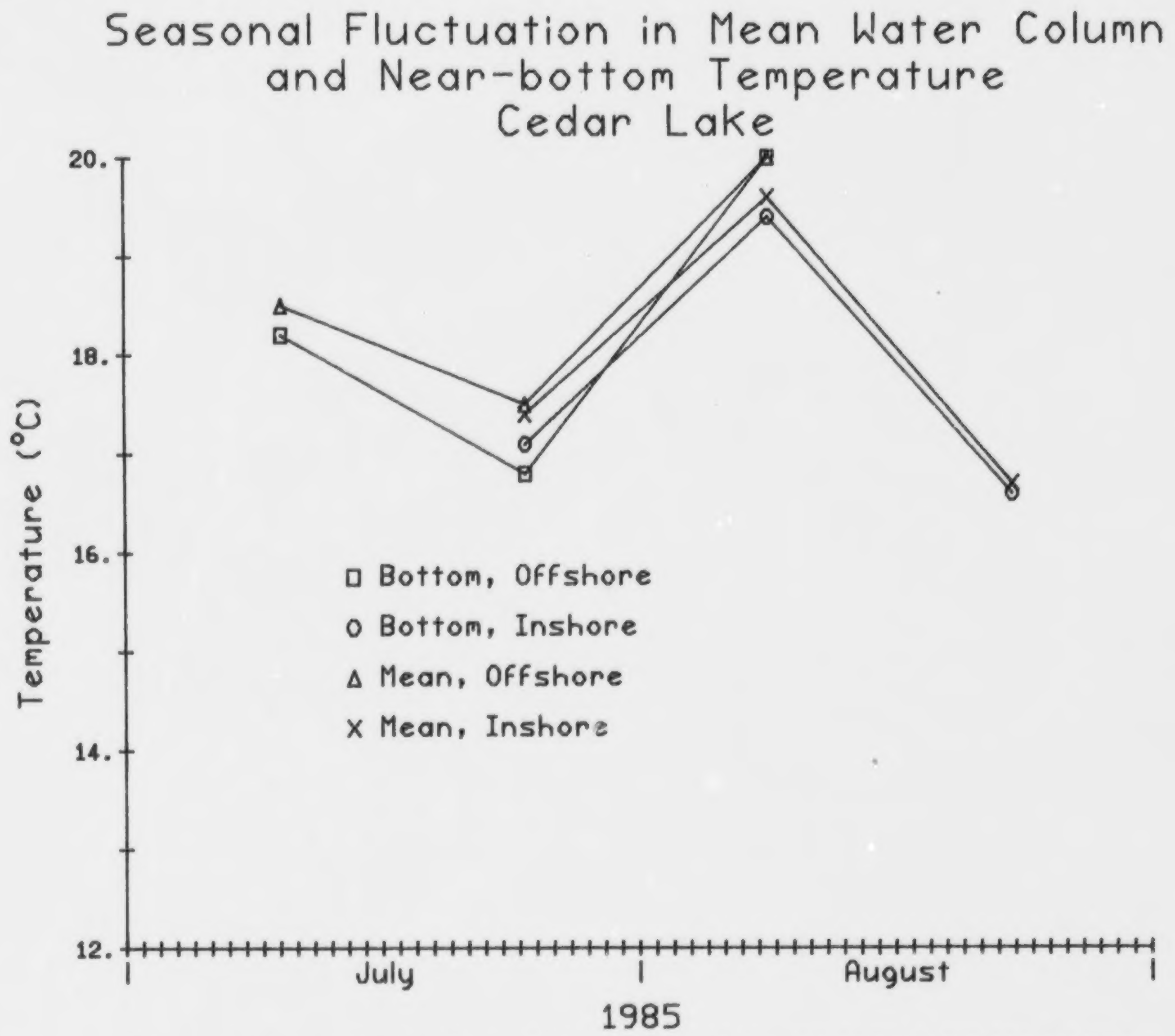


Fig. 4 Seasonal fluctuations of mean water column and near-bottom temperature in Cedar Lake.

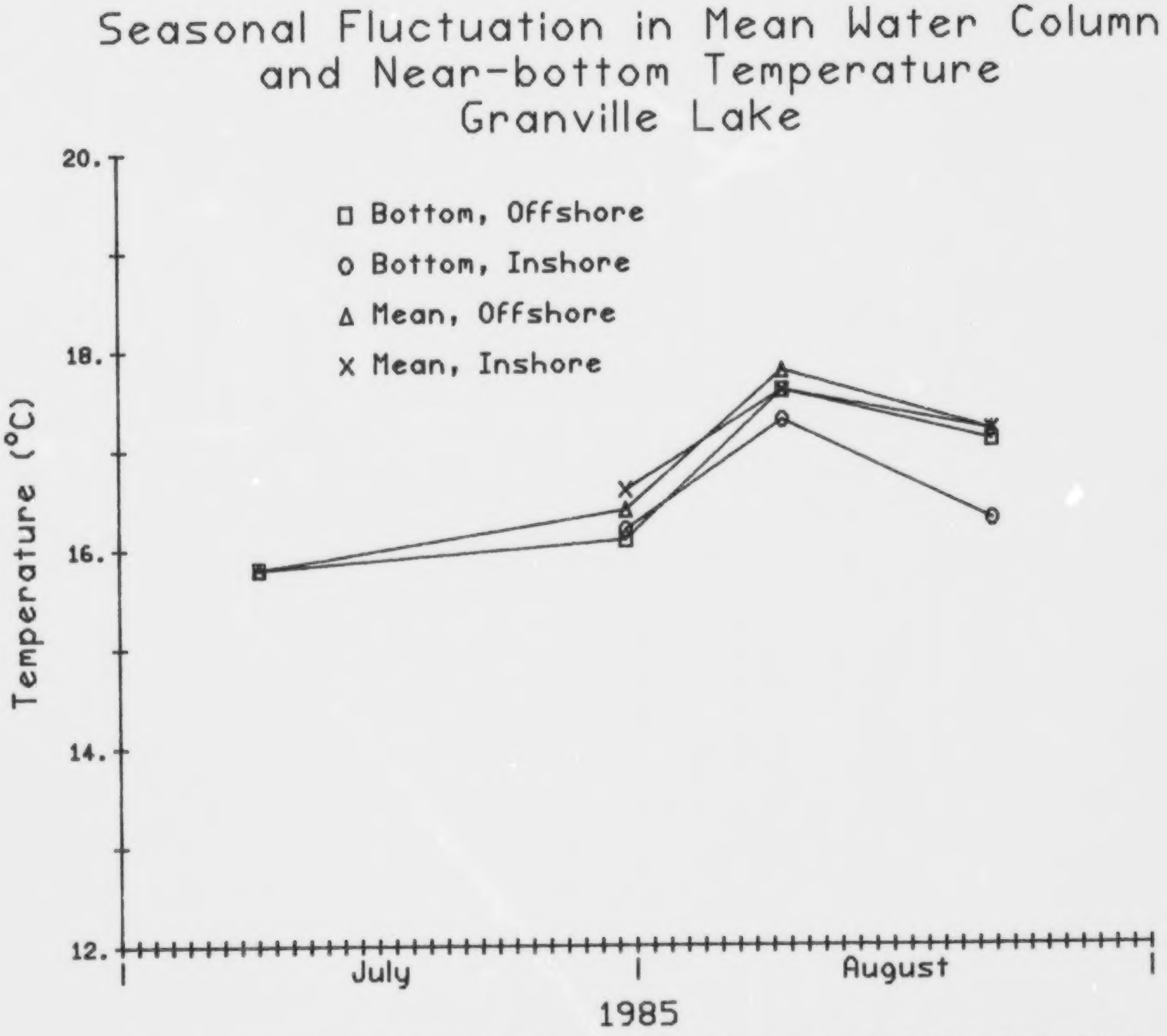


Fig. 5 Seasonal fluctuations of mean water column and near-bottom temperature in Granville Lake.

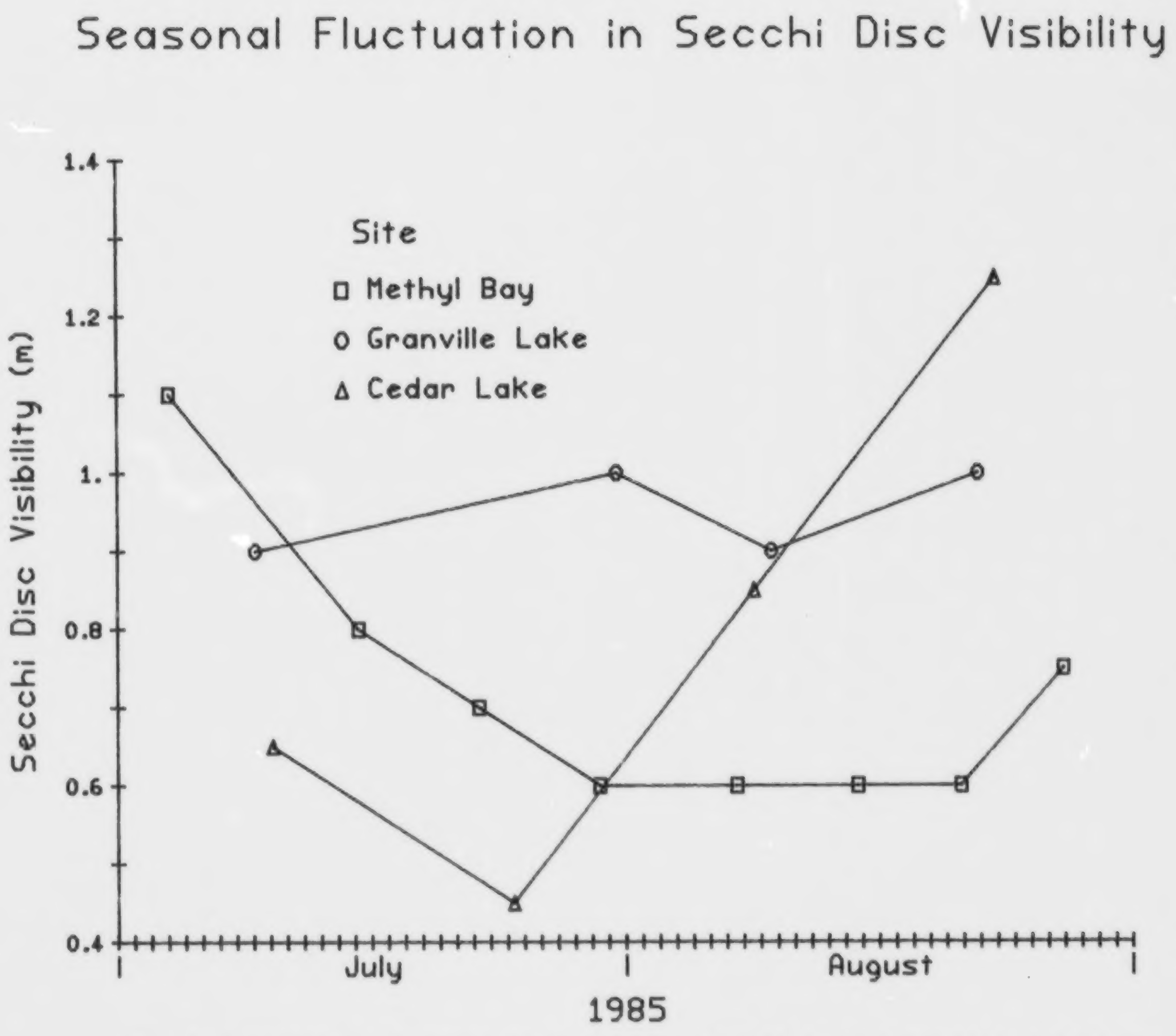


Fig. 6 Seasonal fluctuations in Secchi disk visibility in Cedar Lake, Methyl Bay, and Granville Lake.

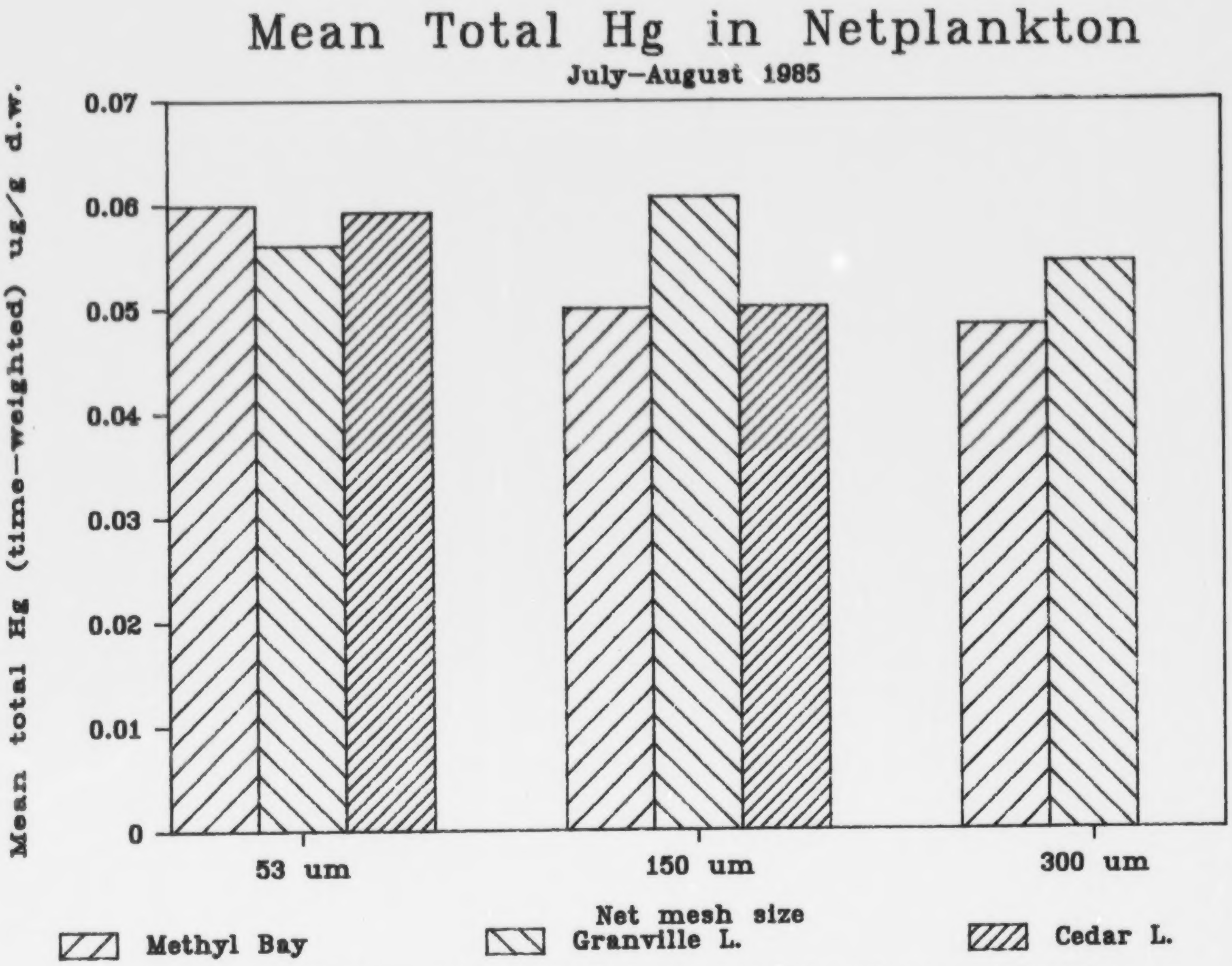


Fig. 7 Seasonal mean (time-weighted) concentrations of total mercury in plankton samples collected with 53, 150, and 300 um mesh-size nets from Granville Lake, Methyl Bay, and Cedar Lake.

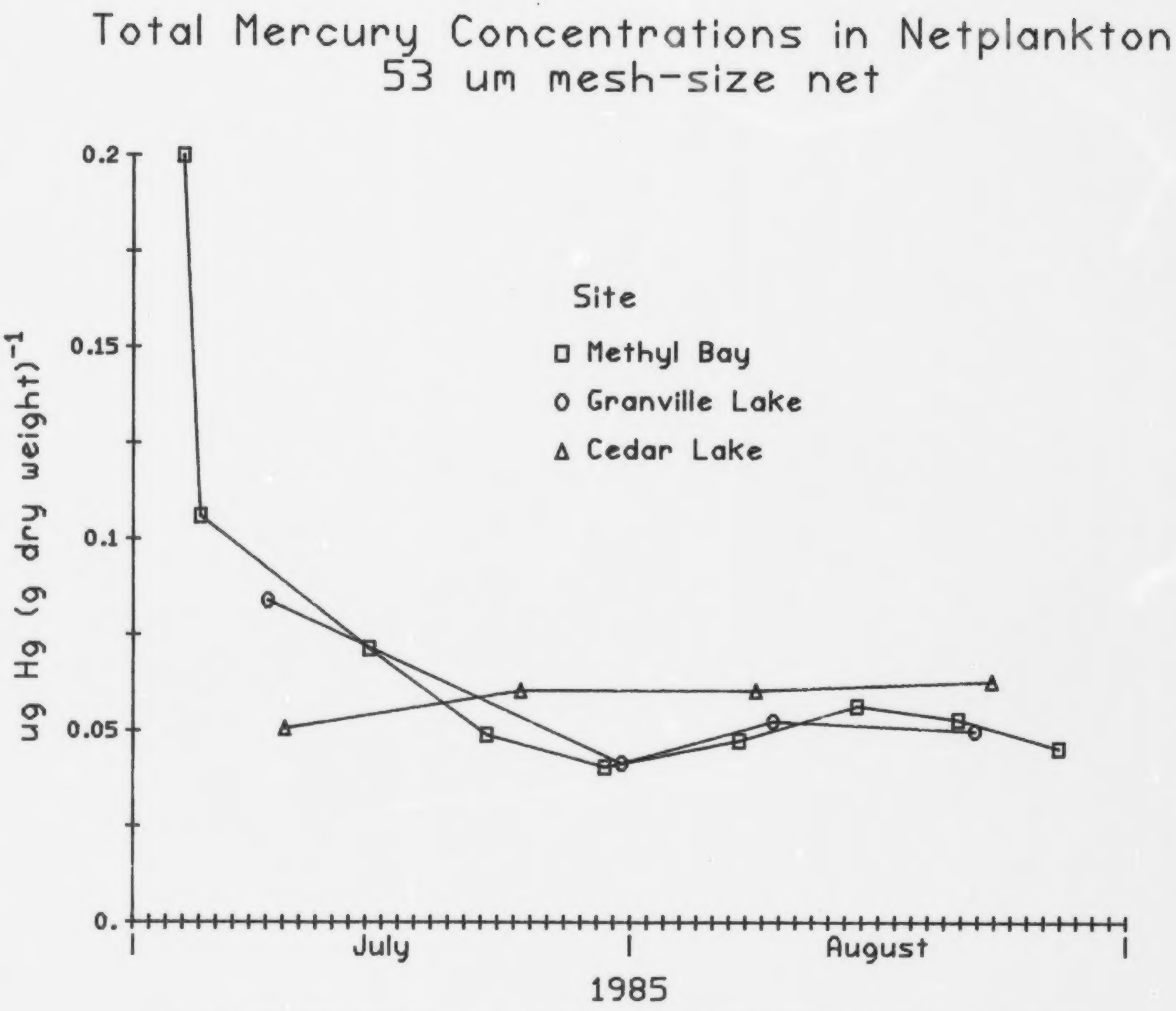
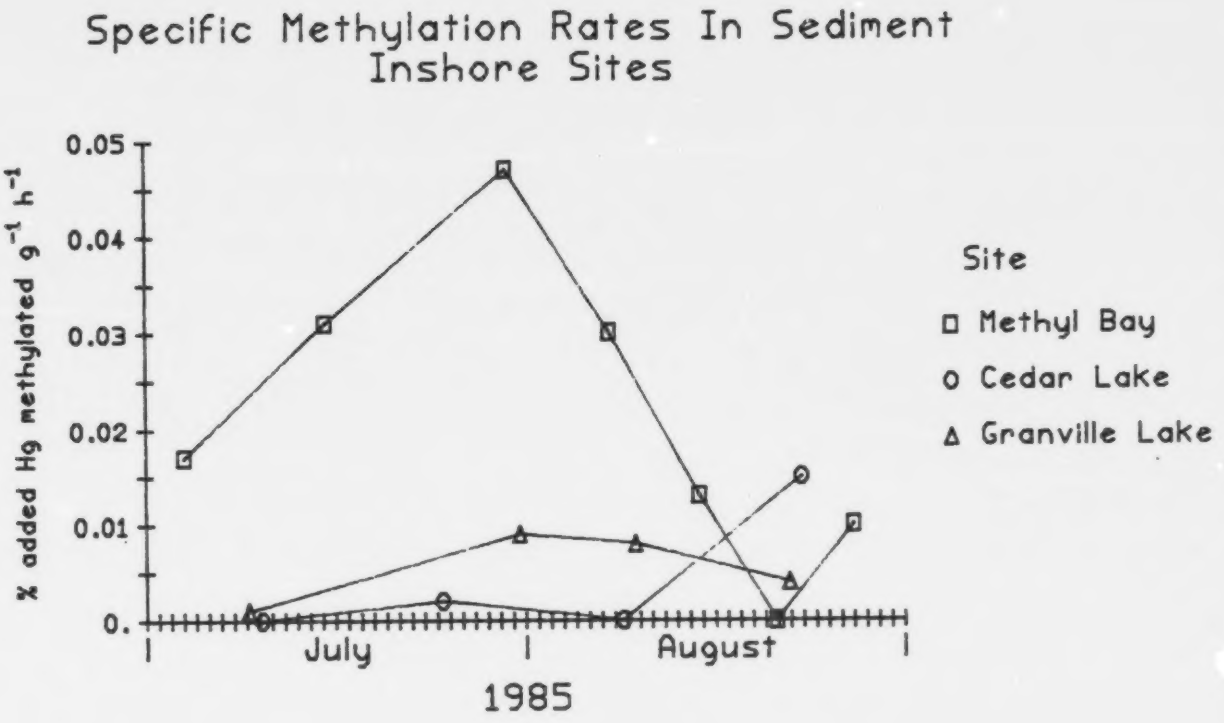


Fig. 8 Seasonal fluctuations in the total mercury concentration in plankton (53 um mesh-size net) from Granville Lake, Methyl Bay, and Cedar Lake.



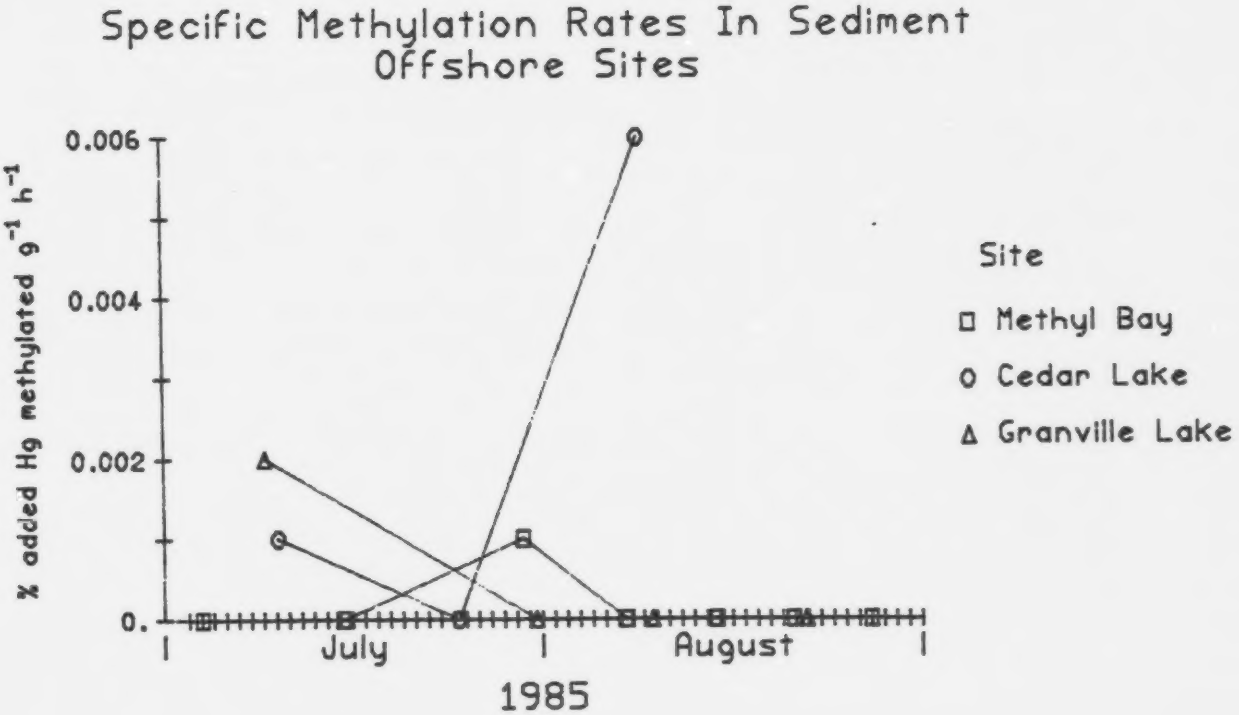
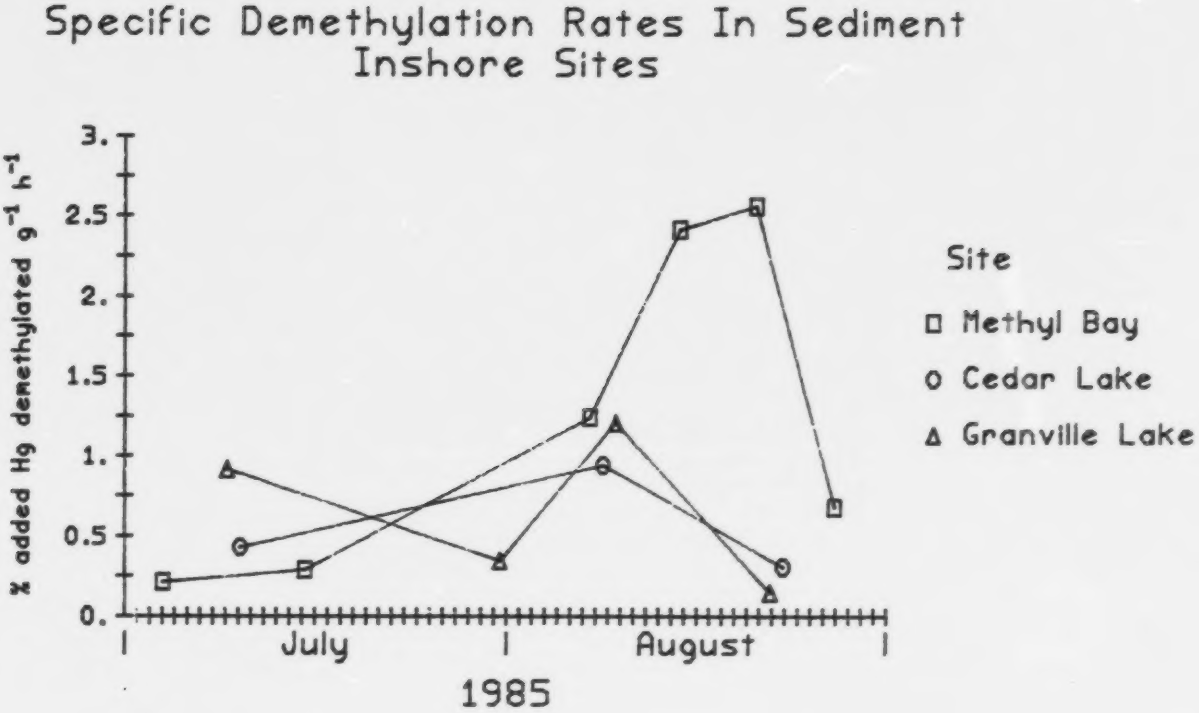


Fig. 9 Seasonal fluctuations in specific rates of mercury methylation; July-August 1985. a) Inshore sites b) Offshore sites.



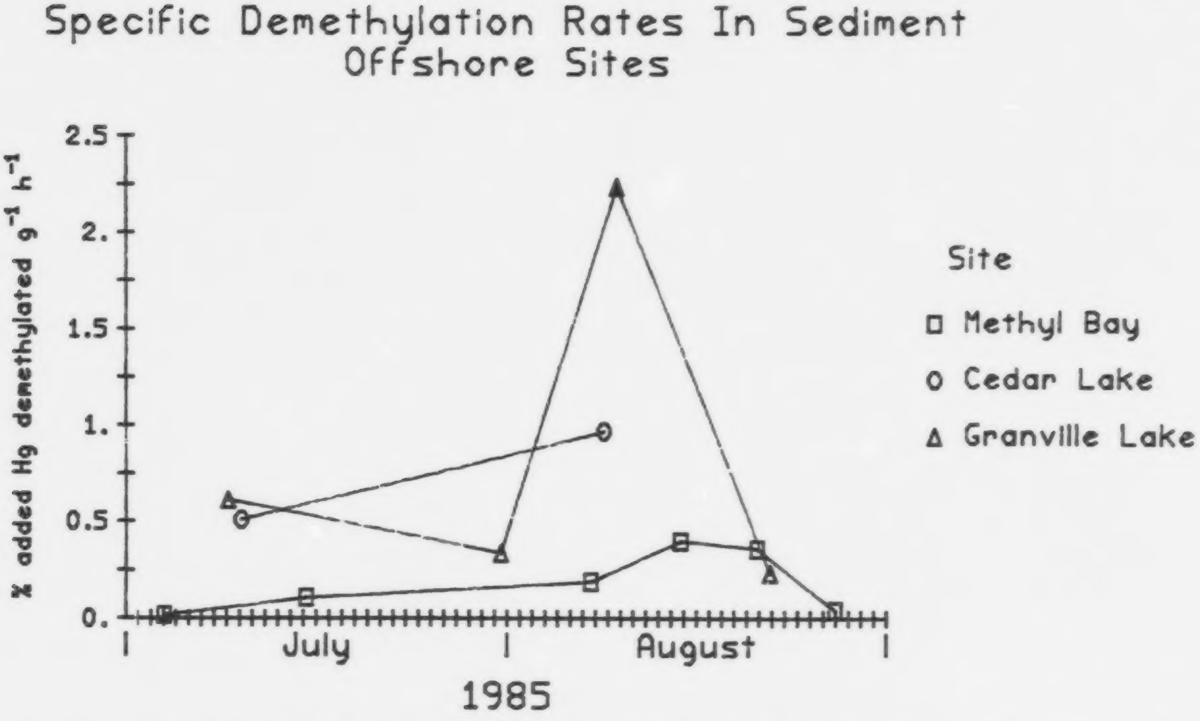
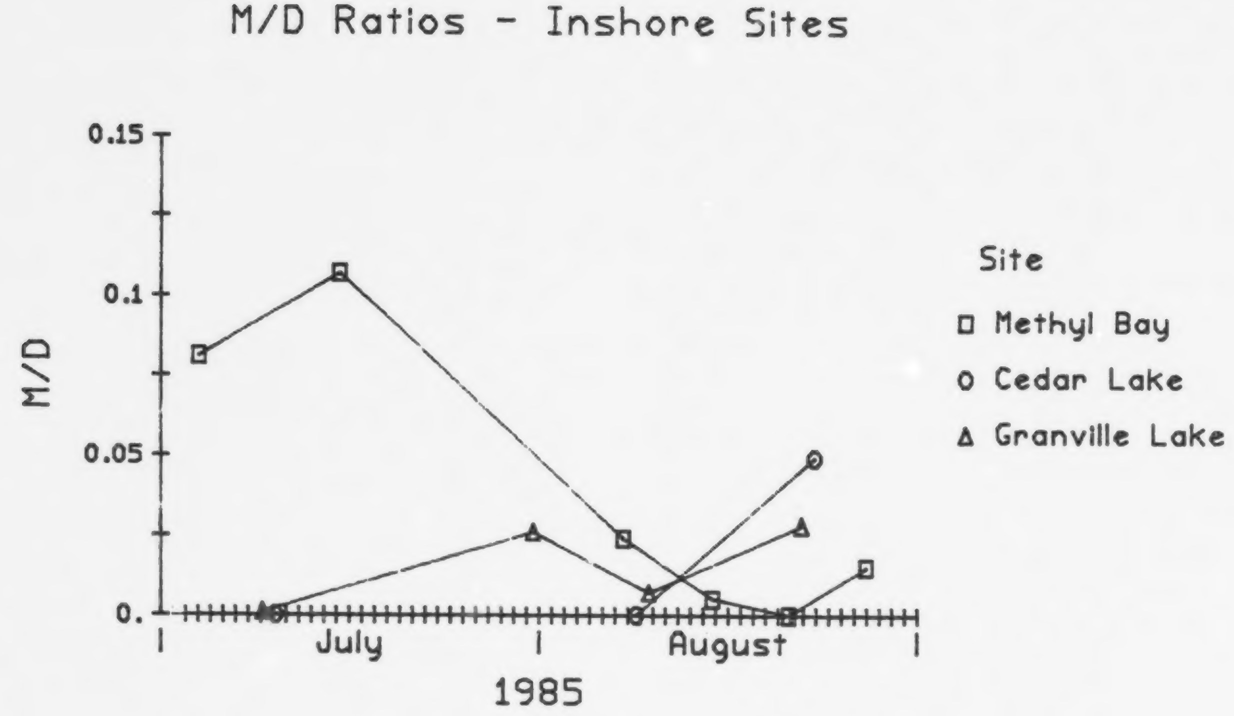


Fig. 10 Seasonal fluctuations in specific rates of methyl mercury degradation (demethylation); July-August 1985.
a) Inshore sites b) Offshore sites.



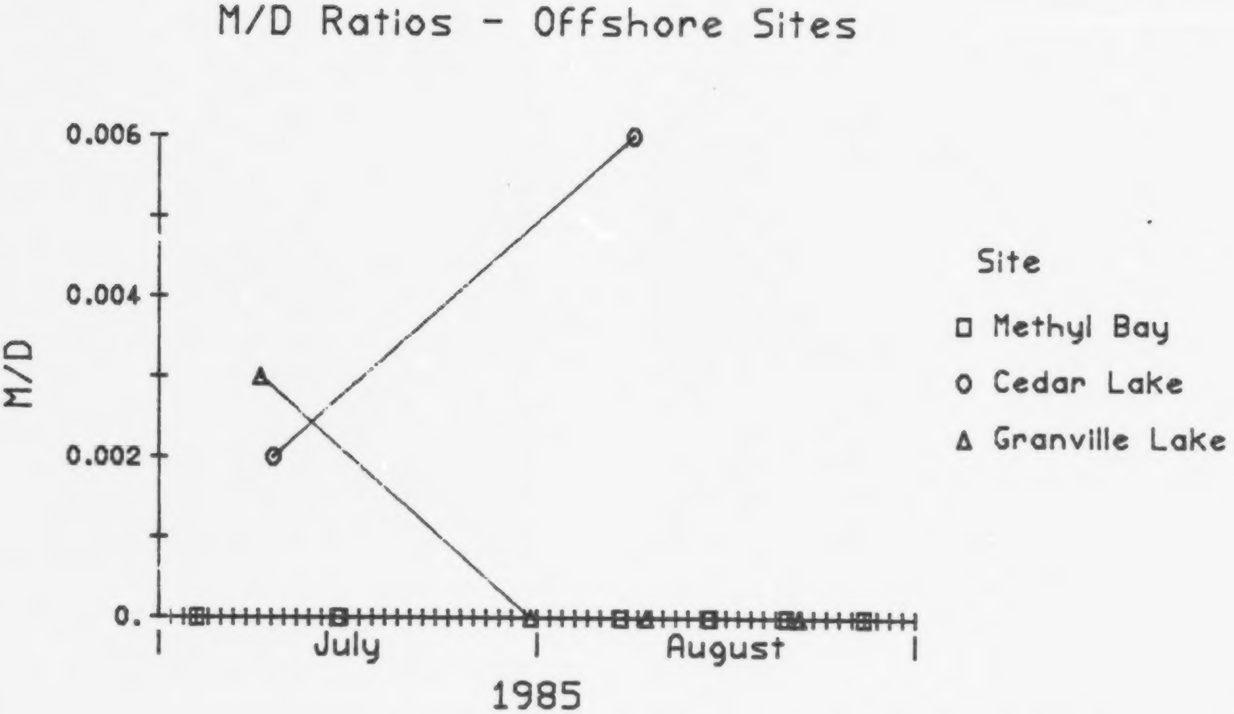


Fig. 11 Seasonal fluctuations in the med ratio; July-August 1985.
a) Inshore sites b) Offshore sites.

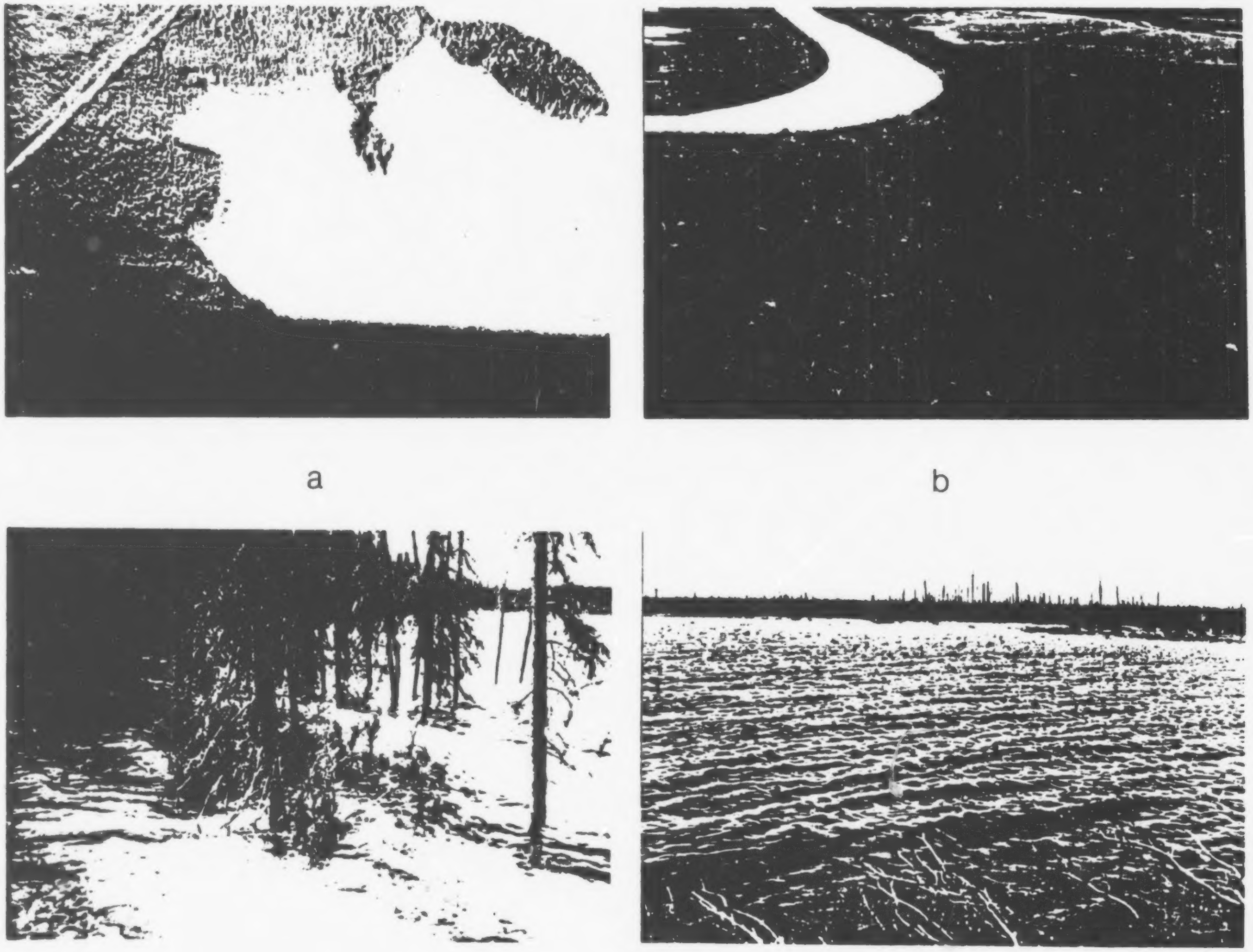


Fig. 12 Photographs of the sampling sites and shorelines in Cedar Lake and Methyl Bay.

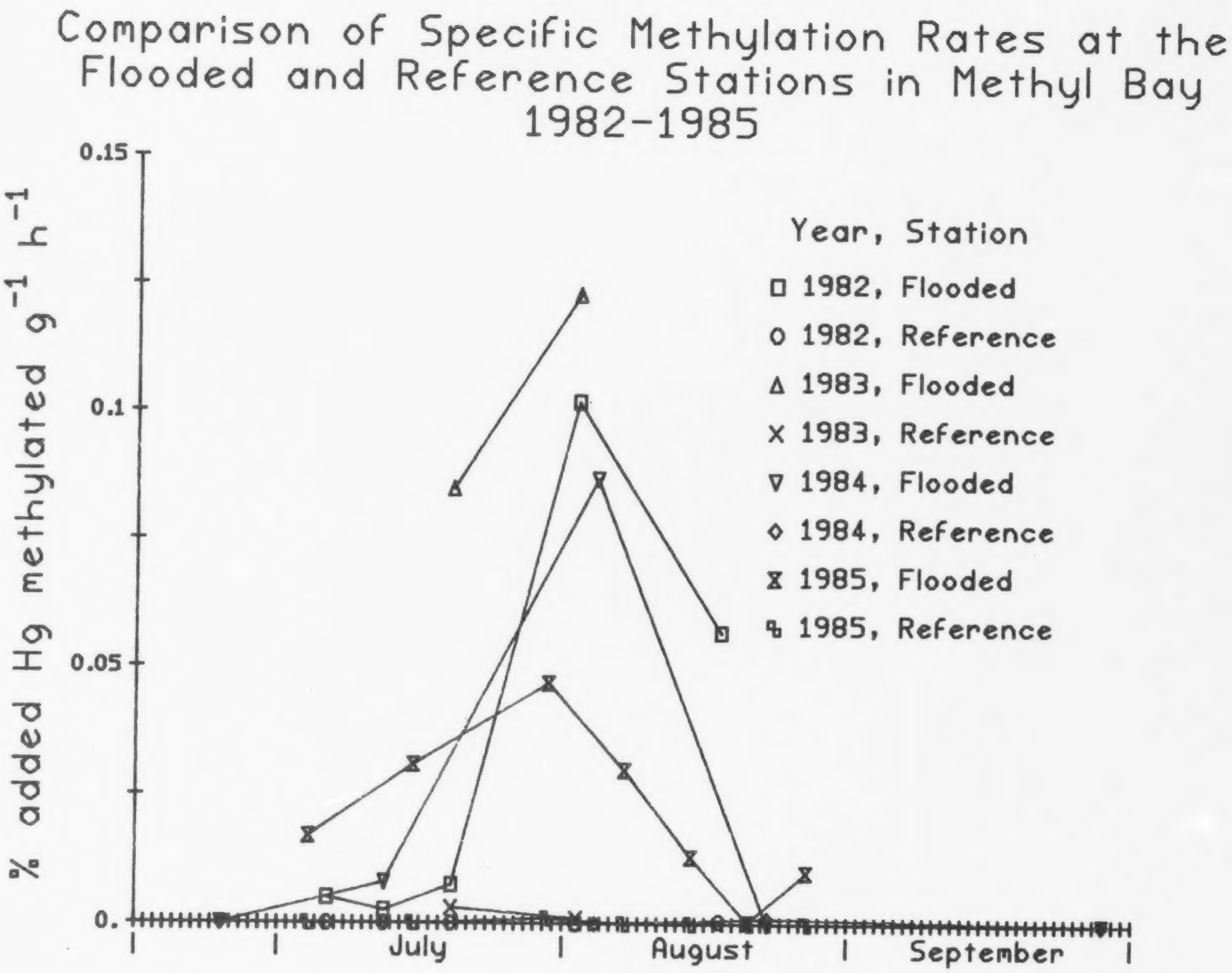


Fig. 13 Comparison of specific methylation rates measured at the flooded and unflooded stations in Methyl Bay; 1982-1984. Data from: Appendix 3, Ramlal et al. 1986b, and this study.

Mercury Survey 1985
Lake: Granville
Site: Campground/Water intake
Summary of physical and chemical measurements

Station	Type	Date	DOY	Station depth (m)	Secchi (m)	Depth (m)	Temp (oC)	Cond (uS/cm)	Oxygen (ppm)	Depth-weighted mean Temp	Depth-weighted mean Cond	Depth-weighted mean Oxygen
1	I-U	9 July 85	31237	0.80	>0.8	NA	NA	NA	NA	NA	NA	NA
2	I-U	9 July 85	31237	1.10	0.90	NA	NA	NA	NA	NA	NA	NA
3	I-U	9 July 85	31237	1.10	0.90	NA	NA	NA	NA	NA	NA	NA
4	O-U	9 July 85	31237	2.50	0.90	0.15	15.80	42.00	NA	15.80	42.00	NA
						0.50	15.80	42.00	NA			
						1.00	15.80	42.00	NA			
						1.50	15.80	42.00	NA			
						2.00	15.80	42.00	NA			
						2.50	15.80	42.00	NA			
1	I-U	30 July 85	31258	1.30	1.00	0.15	17.00	NA	8.60	16.48	NA	7.80
						0.50	16.80	NA	8.00			
						1.00	16.10	NA	7.50			
						1.30	16.00	NA	7.10			
2	I-U	30 July 85	31258	1.30	0.95	0.15	17.10	NA	8.25	16.85	NA	7.95
						0.50	17.00	NA	8.10			
						1.00	16.70	NA	7.90			
						1.30	16.50	NA	7.30			
3	I-U	30 July 85	31258	2.00	1.00	0.15	17.10	NA	8.20	16.48	NA	7.18
						0.50	16.80	NA	7.60			
						1.00	16.50	NA	7.20			
						1.50	16.20	NA	7.00			
						2.00	16.00	NA	6.10			
4	O-U	30 July 85	31258	10.00	1.00	0.15	16.80	NA	8.40	16.39	NA	7.29
						1.00	16.80	NA	8.00			
						2.00	16.70	NA	7.80			
						3.00	16.70	NA	7.70			
						4.00	16.30	NA	7.40			
						5.00	16.20	NA	7.20			
						6.00	16.20	NA	6.90			
						7.00	16.20	NA	6.90			
						8.00	16.20	NA	6.80			
						9.00	16.20	NA	6.90			
						10.00	16.10	NA	6.50			
1	I-U	8 Aug 85	31267	1.60	1.00	0.15	17.90	97.00	7.10	17.33	97.00	6.43
						0.50	17.50	97.00	6.50			
						1.00	17.20	97.00	6.30			
						1.60	17.00	97.00	6.20			
2	I-U	8 Aug 85	31267	1.80	0.90	0.15	18.00	97.00	7.80	17.57	96.11	7.59
						0.50	17.90	96.00	7.70			

Mercury Survey 1985
Lake: Granville
Site: Campground/Water intake
Summary of physical and chemical measurements

Station	Type	Date	DOY	Station depth (m)	Secchi (m)	Depth (m)	Temp (oC)	Cond (uS/cm)	Oxygen (ppm)	Depth-weighted mean		
										Temp	Cond	Oxygen
						1.00	17.70	96.00	7.70			
						1.50	17.10	96.00	7.50			
						1.80	17.00	96.00	6.90			
3	I-U	8 Aug 85	31267	2.50	0.80	0.15	18.00	97.00	NA	17.97	97.00	NA
						0.50	18.00	97.00	NA			
						1.00	18.00	97.00	NA			
						1.50	18.00	97.00	NA			
						2.00	17.90	97.00	NA			
						2.50	17.90	97.00	NA			
4	O-U	8 Aug 85	31267	9.25	0.90	0.15	17.80	97.00	NA	17.75	97.00	NA
						1.00	17.80	97.00	NA			
						2.00	17.80	97.00	NA			
						3.00	17.80	97.00	NA			
						4.00	17.80	97.00	NA			
						5.00	17.80	97.00	NA			
						6.00	17.80	97.00	NA			
						7.00	17.70	97.00	NA			
						8.00	17.60	97.00	NA			
						9.00	17.60	97.00	NA			
						9.25	17.60	97.00	NA			
1	I-U	20 Aug 85	31279	1.80	0.70	0.15	17.60	98.00	9.40	17.24	98.09	8.61
						0.50	17.60	98.00	9.00			
						1.00	17.40	98.00	8.40			
						1.50	16.90	98.00	8.60			
						1.80	16.20	99.00	7.30			
2	I-U	20 Aug 85	31279	2.40	0.90	0.15	17.40	99.00	9.10	17.15	99.00	8.18
						0.50	17.40	99.00	8.50			
						1.00	17.20	99.00	8.40			
						1.50	17.20	99.00	8.10			
						2.00	17.00	99.00	8.00			
						2.40	16.50	99.00	6.70			
3	I-U	20 Aug 85	31279	1.90	1.10	0.15	17.60	98.00	8.10	17.33	98.00	7.51
						0.50	17.60	98.00	7.60			
						1.00	17.50	98.00	7.50			
						1.50	17.30	98.00	7.30			
						1.90	16.20	98.00	7.30			
4	O-U	20 Aug 85	31279	9.00	1.00	0.15	17.50	98.00	7.00	17.22	98.90	7.10
						1.00	17.30	98.00	7.00			
						2.00	17.30	98.00	7.00			
						3.00	17.20	98.00	6.70			
						4.00	17.20	99.00	7.00			
						5.00	17.20	99.00	7.10			

Mercury Survey 1985
Lake: Granville
Site: Campground/Water intake
Summary of physical and chemical measurements

Station	Type	Date	DOY	Station depth (m)	Secchi (m)	Depth (m)	Temp (oC)	Cond (uS/cm)	Oxygen (ppm)	Depth-weighted mean Temp	Depth-weighted mean Cond	Depth-weighted mean Oxygen
						6.00	17.20	99.00	7.30			
						7.00	17.20	100.00	7.30			
						8.00	17.10	100.00	7.30			
						9.00	17.10	100.00	7.30			

Mercury Survey 1985
Lake: Southern Indian
Site: Methyl Bay
Summary of physical and chemical measurements

Station	Type	Date	DOY	Station depth (m)	Secchi (m)	Depth (m)	Temp (oC)	Cond (uS/cm)	Oxygen (ppm)	Depth-weighted mean Temp	Depth-weighted mean Cond	Depth-weighted mean Oxygen
1	I-F	4 July 85	31232	1.80	0.90	0.15	17.80	NA	9.50	17.21	NA	9.18
						0.50	17.50	NA	9.40			
						1.00	17.00	NA	9.10			
						1.50	17.00	NA	9.00			
						1.80	17.00	NA	8.90			
2	I-F	4 July 85	31232	1.50	1.00	0.15	18.00	NA	10.80	17.94	NA	9.73
						0.50	18.00	NA	10.10	Note: Ox	meas	suspect
						1.00	17.90	NA	9.40			
						1.50	17.90	NA	9.00			
3	I-F	4 July 85	31232	1.90	1.00	0.15	18.10	NA	12.50	17.95	NA	11.44
						0.50	18.00	NA	12.20	Note: Ox	meas	suspect
						1.00	18.00	NA	11.20			
						1.50	18.00	NA	11.00			
						1.90	17.50	NA	10.50			
4	O-U	4 July 85	31232	6.75	1.10	0.15	17.70	67.00	NA	15.45	66.09	NA
						0.50	17.20	67.00	NA			
						1.00	17.10	67.00	NA			
						1.50	17.10	65.00	NA			
						2.00	16.60	66.00	NA			
						2.50	16.30	66.00	NA			
						3.00	16.10	66.00	NA			
						3.50	15.50	66.00	NA			
						4.00	15.00	66.00	NA			
						4.50	14.50	66.00	NA			
						5.00	14.10	66.00	NA			
						5.50	13.50	66.00	NA			
						6.00	13.20	66.00	NA			
						6.50	12.90	66.00	NA			
4	O-U	5 July 85	31233	6.50		0.15	19.00	NA	9.40	16.41	NA	9.07
						0.50	18.50	NA	9.40			
						1.00	18.00	NA	9.20			
						1.50	17.50	NA	9.30			
						2.00	17.50	NA	9.10			
						2.50	17.00	NA	9.40			
						3.00	16.50	NA	9.00			
						3.50	16.00	NA	9.50			
						4.00	16.00	NA	9.40			
						4.50	15.50	NA	9.40			
						5.00	15.50	NA	9.00			
						5.50	15.00	NA	9.10			
						6.00	14.50	NA	8.90			
						6.50	14.00	NA	5.10			
1	I-F	15 July 85	31243	1.40	0.60	0.15	16.00	64.00	8.40	16.00	63.30	8.17

Mercury Survey 1985
Lake: Southern Indian
Site: Methyl Bay
Summary of physical and chemical measurements

Station	Type	Date	DOY	Station depth (m)	Secchi (m)	Depth (m)	Temp (oC)	Cond (uS/cm)	Oxygen (ppm)	Depth-weighted mean		
										Temp	Cond	Oxygen
						0.50	16.00	63.00	8.20			
						1.00	16.00	63.00	8.00			
						1.40	16.00	64.00	8.30			
2	I-F	15 July 85	31243	1.70	1.00	0.15	15.30	64.00	9.10	15.15	64.06	8.42
						0.50	15.20	64.00	8.90			
						1.00	15.20	64.00	8.60			
						1.50	15.00	64.00	7.50			
						1.70	15.00	65.00	7.50			
3	I-F	15 July 85	31243	1.90	0.90	0.15	16.00	64.00	8.20	15.19	64.00	7.84
						0.50	15.50	64.00	8.00			
						1.00	15.00	64.00	7.60			
						1.50	14.50	64.00	7.80			
4	O-U	15 July 85	31243	7.00	0.80	0.15	15.90	66.00	8.10	15.43	64.76	8.00
						0.50	15.90	66.00	8.10			
						1.00	15.90	65.00	8.10			
						1.50	15.80	65.00	8.20			
						2.00	15.80	65.00	8.10			
						2.50	15.80	65.00	8.20			
						3.00	15.50	65.00	8.20			
						3.50	15.40	65.00	8.00			
						4.00	15.30	65.00	7.80			
						4.50	15.20	65.00	7.90			
						5.00	15.10	64.00	7.90			
						5.50	15.00	64.00	7.90			
						6.00	15.00	64.00	7.90			
						6.50	15.00	64.00	7.80			
						7.00	15.00	64.00	7.70			
1	I-F	22 July 85	31250	1.30	0.70	0.15	16.00	64.00	8.20	15.92	64.00	7.25
						0.50	15.90	64.00	8.20			
						1.00	15.90	64.00	7.80			
						1.30	15.90	64.00	2.00			
2	I-F	22 July 85	31250	1.80	0.80	0.15	15.50	63.00	8.20	15.22	63.00	7.43
						0.50	15.40	63.00	8.20			
						1.00	15.20	63.00	7.20			
						1.50	15.00	63.00	7.10			
						1.80	15.00	63.00	6.00			
3	I-F	22 July 85	31250	2.80	0.80	0.15	15.40	66.00	8.20	14.32	80.95	5.57
						0.50	15.30	67.00	7.30			
						1.00	15.10	70.00	7.10			
						1.50	15.00	90.00	6.70			
						2.00	13.40	90.00	5.20			
						2.50	12.80	90.00	1.40			

Mercury Survey 1985
Lake: Southern Indian
Site: Methyl Bay
Summary of physical and chemical measurements

Station	Type	Date	DOY	Station depth (m)	Secchi (m)	Depth (m)	Temp (oC)	Cond (uS/cm)	Oxygen (ppm)	Depth-weighted mean		
										Temp	Cond	Oxygen
						2.80	12.60	90.00	1.00			
4	O-U	22 July 85	31250	7.00	0.70	0.15	15.00	65.00	NA	14.81	64.31	NA
						0.50	15.00	65.00	NA			
						1.00	15.00	65.00	NA			
						1.50	15.00	65.00	NA			
						2.00	15.00	65.00	NA			
						2.50	15.00	64.00	NA			
						3.00	15.00	64.00	NA			
						3.50	15.00	64.00	NA			
						4.00	14.83	64.00	NA			
						4.50	14.60	64.00	NA			
						5.00	14.60	64.00	NA			
						5.50	14.58	64.00	NA			
						6.00	14.52	64.00	NA			
						6.50	14.50	64.00	NA			
						7.00	14.48	64.00	NA			
1	I-F	29 July 85	31257	1.40	0.60	0.15	15.70	66.00	9.00	15.63	66.00	8.36
						0.50	15.70	66.00	9.00			
						1.00	15.60	66.00	8.20			
						1.40	15.50	66.00	6.80			
2	I-F	29 July 85	31257	1.55	0.90	0.15	16.00	66.00	8.90	16.00	66.57	7.94
						0.50	16.00	66.00	8.20			
						1.00	16.00	67.00	7.60			
						1.55	16.00	67.00	7.60			
3	I-F	29 July 85	31257	0.70	>.7	0.15	16.00	69.00	7.10	16.00	69.00	7.10
						0.50	16.00	69.00	7.10			
4	O-U	29 July 85	31257	7.00	0.60	0.15	15.20	65.00	8.40	14.85	66.33	7.64
						0.50	15.20	66.00	8.00			
						1.00	15.20	66.00	8.00			
						1.50	15.10	66.00	8.00			
						2.00	15.10	66.00	7.80			
						2.50	15.10	66.00	7.80			
						3.00	15.10	66.00	7.70			
						3.50	15.10	66.00	7.80			
						4.00	15.00	66.00	7.70			
						4.50	14.90	66.00	7.50			
						5.00	14.50	67.00	7.40			
						5.50	14.20	67.00	7.10			
						6.00	14.00	67.00	7.00			
						6.50	14.00	69.00	7.00			
1	I-F	6 Aug 85	31265	1.40	0.50	0.15	16.80	101.00	7.20	16.70	100.84	6.39
						0.50	16.70	101.00	6.60			

Mercury Survey 1985
Lake: Southern Indian
Site: Methyl Bay
Summary of physical and chemical measurements

Station	Type	Date	DOY	Station depth (m)	Secchi (m)	Depth (m)	Temp (oC)	Cond (uS/cm)	Oxygen (ppm)	Depth-weighted mean Temp	Depth-weighted mean Cond	Depth-weighted mean Oxygen
						1.00	16.70	101.00	6.10			
						1.40	16.60	100.00	5.90			
2	I-F	6 Aug 85	31265	1.00	0.65	0.15	17.00	101.00	6.00	17.00	100.85	5.88
						0.50	17.00	101.00	6.00			
						0.65	17.00	100.00	5.20			
3	I-F	6 Aug 85	31265	0.75	0.60	0.15	16.50	103.00	7.90	16.20	103.00	6.47
						0.50	16.10	103.00	6.20			
						0.75	16.00	103.00	5.10			
4	O-U	6 Aug 85	31265	7.00	0.60	0.15	16.80	102.00	7.40	16.15	103.89	6.90
						1.00	16.60	103.00	7.30			
						2.00	16.50	103.00	7.20			
						3.00	16.10	103.00	7.00			
						4.00	16.00	104.00	7.10			
						5.00	16.00	105.00	7.00			
						6.00	15.90	105.00	6.80			
						7.00	15.20	106.00	4.60			
1	I-F	13 Aug 85	31272	1.40	0.60	0.15	16.00	100.00	9.20	15.98	100.00	8.47
						0.50	16.00	100.00	8.50			
						1.00	16.00	100.00	8.50			
						1.40	15.90	100.00	7.70			
2	I-F	13 Aug 85	31272	0.80	0.75	0.15	16.00	100.00	8.60	16.00	100.00	7.43
						0.50	16.00	100.00	7.50			
						0.80	16.00	100.00	5.90			
3	I-F	13 Aug 85	31272	0.95	0.75	0.15	16.20	100.00	9.10	16.11	100.00	8.00
						0.50	16.10	100.00	8.10			
						0.75	16.00	100.00	6.20			
4	O-U	13 Aug 85	31272	6.85	0.60	0.15	16.10	100.00	8.80	15.80	101.64	8.08
						1.00	16.10	100.00	8.60			
						2.00	16.00	100.00	8.50			
						3.00	16.00	101.00	8.40			
						4.00	16.00	101.00	8.30			
						5.00	16.00	103.00	8.40			
						6.00	15.00	104.00	7.70			
						6.90	14.80	105.00	4.30			
1	I-F	19 Aug 85	31278	1.00	0.70	0.15	16.10	102.00	8.20	16.02	100.41	7.57
						0.50	16.00	100.00	7.70			
						1.00	16.00	100.00	6.90			
2	I-F	19 Aug 85	31278	0.80	0.75	0.15	16.10	100.00	8.60	16.03	100.00	7.89
						0.50	16.00	100.00	8.10			

Mercury Survey 1985
Lake: Southern Indian
Site: Methyl Bay
Summary of physical and chemical measurements

Station	Type	Date	DOY	Station depth (m)	Secchi (m)	Depth (m)	Temp (oC)	Cond (uS/cm)	Oxygen (ppm)	Depth-weighted mean		
										Temp	Cond	Oxygen
						0.80	16.00	100.00	6.60			
3	I-F	19 Aug 85	31278	0.80	>.8	0.15	16.10	102.00	7.20	16.08	102.00	7.27
						0.50	16.10	102.00	7.30			
						0.80	16.00	102.00	7.30			
4	O-U	19 Aug 85	31278	6.75	0.60	0.15	16.30	101.00	8.70	15.71	102.63	8.19
						1.00	16.30	101.00	8.50			
						2.00	16.10	101.00	8.60			
						3.00	15.90	103.00	8.40			
						4.00	15.50	103.00	8.20			
						5.00	15.30	104.00	8.10			
						6.00	15.10	104.00	7.80			
						6.75	15.10	104.00	6.30			
1	I-F	25 Aug 85	31284	1.30	0.60	0.15	17.10	NA	8.60	17.02	NA	7.21
						0.50	17.00	NA	7.30			
						1.00	17.00	NA	6.80			
						1.30	17.00	NA	6.40			
2	I-F	25 Aug 85	31284	0.80	0.70	0.15	17.30	NA	5.90	17.13	NA	5.65
						0.50	17.10	NA	5.90			
						0.80	17.00	NA	4.80			
3	I-F	25 Aug 85	31284	0.60	>.6	0.15	17.00	NA	6.70	16.86	NA	5.18
						0.50	16.80	NA	4.10			
						0.60	16.60	NA	4.70			
4	O-U	25 Aug 85	31284	6.75	0.75	0.15	17.30	NA	8.20	16.33	NA	7.42
						1.00	17.10	NA	8.10			
						2.00	16.90	NA	8.10			
						3.00	16.80	NA	8.00			
						4.00	16.10	NA	7.70			
						5.00	16.00	NA	7.40			
						6.00	15.10	NA	7.00			
						6.75	14.90	NA	1.70			

Mercury Survey 1985
Lake: Cedar
Summary of physical and chemical measurements

Station	Type	Date	DOY	Station depth (m)	Secchi (m)	Depth (m)	Temp (oC)	Cond (uS/cm)	Oxygen (ppm)	Depth-weighted mean		
										Temp	Cond	Oxygen
1	O-F	10 July 85	31238	2.00	0.60	0.15	18.00	310.00	9.00	18.00	310.00	8.71
						0.50	18.00	310.00	8.90			
						1.00	18.00	310.00	8.60			
						1.50	18.00	310.00	8.60			
						2.00	18.00	310.00	8.60			
2	I-F	10 July 85	31238	1.10	0.60	NA	NA	NA	NA	NA	NA	NA
3	I-F	10 July 85	31238	1.00	NA	NA	NA	NA	NA	NA	NA	NA
4	I-F	10 July 85	31238	1.40	NA	NA	NA	NA	NA	NA	NA	NA
5	I-F	10 July 85	31238	1.40	NA	NA	NA	NA	NA	NA	NA	NA
6	O-F	10 July 85	31238	1.80	0.70	0.15	19.10	NA	9.40	18.99	NA	8.83
						0.50	19.10	NA	9.00			
						1.00	19.10	NA	8.70			
						1.50	18.90	NA	8.70			
						1.80	18.40	NA	8.50			
1	O-F	24 July 85	31252	1.90	0.50	0.15	18.00	NA	9.30	17.49	NA	8.83
						0.50	18.00	NA	9.10			
						1.00	17.90	NA	8.60			
						1.50	17.00	NA	8.70			
						1.90	16.00	NA	8.70			
2	O-F	24 July 85	31252	2.00	0.40	0.15	17.50	NA	9.30	17.50	NA	8.63
						0.50	17.50	NA	8.80			
						1.00	17.50	NA	8.80			
						1.50	17.50	NA	8.60			
						2.00	17.50	NA	7.60			
3	I-F	24 July 85	31252	1.40	>1.4	0.15	17.50	NA	8.19	16.98	NA	7.92
						0.50	17.50	NA	8.00			
						1.00	16.50	NA	7.80			
						1.40	16.50	NA	7.80			
4	I-F	24 July 85	31252	1.50	0.50	NA	NA	NA	NA	NA	NA	NA
5	I-F	24 July 85	31252	1.30	1.00	0.15	17.75	NA	6.70	17.75	NA	6.23
						0.50	17.75	NA	6.30			
						1.00	17.75	NA	6.10			
						1.30	17.75	NA	5.80			
1	O-F	7 Aug 85	31266	2.00	1.20	0.15	20.00	95.00	6.60	20.00	94.86	6.39
						0.50	20.00	95.00	6.50			
						1.00	20.00	95.00	6.40			
						1.50	20.00	95.00	6.30			
						2.00	20.00	94.00	6.20			

Mercury Survey 1985
Lake: Cedar
Summary of physical and chemical measurements

Station	Type	Date	DOY	Station depth (m)	Secchi (m)	Depth (m)	Temp (oC)	Cond (uS/cm)	Oxygen (ppm)	Depth-weighted mean		
										Temp	Cond	Oxygen
2	O-F	7 Aug 85	31266	2.80	0.50	0.15	20.00	95.00	NA	20.00	94.85	NA
						0.50	20.00	95.00	NA			
						1.00	20.00	95.00	NA			
						1.50	20.00	95.00	NA			
						2.00	20.00	95.00	NA			
						2.80	20.00	94.00	NA			
3	I-F	7 Aug 85	31266	1.40	1.30	0.15	19.20	96.00	NA	19.20	96.00	NA
						0.50	19.20	96.00	NA			
						1.00	19.20	96.00	NA			
						1.30	19.20	96.00	NA			
4	I-F	7 Aug 85	31266	1.20	>1.2	0.15	19.40	96.00	NA	19.46	96.00	NA
						0.50	19.50	96.00	NA			
						1.00	19.50	96.00	NA			
						1.20	19.30	96.00	NA			
5	I-F	7 Aug 85	31266	1.20	>1.2	0.15	19.80	96.00	NA	19.70	96.00	NA
						0.50	19.80	96.00	NA			
						1.00	19.60	96.00	NA			
						1.20	19.40	96.00	NA			
6	I-F	7 Aug 85	31266	1.10	0.80	0.15	20.00	95.00	NA	19.94	95.00	NA
						0.50	20.00	95.00	NA			
						1.00	19.80	95.00	NA			
1	I-F	21 Aug 85	31280	1.40	>1.4	0.15	16.50	103.00	6.40	16.45	103.00	6.36
						0.50	16.50	103.00	6.40			
						1.00	16.50	103.00	6.10			
						1.40	16.20	103.00	6.80			
2	I-F	21 Aug 85	31280	1.40	1.15	0.15	16.60	102.00	9.10	16.58	102.00	8.44
						0.50	16.60	102.00	8.70			
						1.00	16.60	102.00	8.50			
						1.40	16.50	102.00	7.20			
3	I-F	21 Aug 85	31280	1.25	>1.25	0.15	16.80	103.00	9.20	16.80	103.00	8.73
						0.50	16.80	103.00	8.80			
						1.00	16.80	103.00	8.60			
						1.25	16.80	103.00	8.20			
4	I-F	21 Aug 85	31280	1.50	1.30	0.15	16.80	103.00	8.80	16.80	103.00	8.39
						0.50	16.80	103.00	8.50			
						1.00	16.80	103.00	8.40			
						1.50	16.80	103.00	7.90			
5	I-F	21 Aug 85	31280	1.40	>1.4	0.15	16.90	103.00	9.20	16.82	103.00	8.52
						0.50	16.90	103.00	9.00			

Mercury Survey 1985
Lake: Cedar
Summary of physical and chemical measurements

Station	Type	Date	DOY	Station depth (m)	Secchi (m)	Depth (m)	Temp (oC)	Cond (uS/cm)	Oxygen (ppm)	Depth-weighted mean Temp	Depth-weighted mean Cond	Depth-weighted mean Oxygen
						1.00	16.80	103.00	8.60			
						1.40	16.60	103.00	6.70			
6	I-F	21 Aug 85	31280	1.80	1.00	0.15	16.90	102.00	8.70	16.90	102.00	8.21
						0.50	16.90	102.00	8.20			
						1.00	16.90	102.00	8.20			
						1.50	16.90	102.00	8.20			
						1.80	16.90	102.00	7.70			

SPECIAL NOTE

This report was prepared for the Canada-Manitoba Agreement on the Study and Monitoring of Mercury in the Churchill River Diversion. Any conclusions, recommendations, or opinions expressed herein are those of the author and are not necessarily the same as those of the Steering Committee for this Agreement.

APPENDIX 5

METHODS AND RESULTS FOR MERCURY IN LARGE FISH FROM SOUTHERN INDIAN LAKE AND ISSETT LAKE, 1975-85

by

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INTRODUCTION

This appendix will describe the detailed methods used to sample and determine mercury concentrations in large fish species in Southern Indian Lake and Issett Lake, Manitoba. Sampling was conducted over the period 1975-85 with the most detailed results available for 1982, 1983 and 1984. Results of laboratory and statistical analyses will be presented in tables and figures included with this appendix.

METHODS AND MATERIALS

Samples of large fish were collected in various years within five areas of Southern Indian Lake and at Issett Lake. The following six species were sampled: lake whitefish (Coregonus clupeaformis), walleye (Stizostedion vitreum), northern pike (Esox lucius), cisco (Coregonus artedii and C. nigripinnis), longnose sucker (Catostomus catostomus), and burbot (Lota lota). The five areas sampled on Southern Indian Lake were The Channel (between Regions 2 and 6), Camp 9 (Region 2), Region 4, Region 6 (South Bay), and Region 5 (Fig. 1). The number of fish sampled from each area in each year can be seen in Tables 1-4. Fish were captured with gangs of graded mesh gill nets. Meshes used were usually 8.9, 10.8 and 13.3 cm stretched measure but occasionally included meshes as small as 3.8 cm stretched measure. Most fish which were smaller than those commonly caught in gill nets of 8.9 cm stretched mesh were excluded from analysis. Fork length (to the nearest mm) and round weight (to the nearest 25 g) were determined for all fish (except in 1985; see below). A muscle sample was removed and kept frozen until analysis. Mercury concentrations were determined using the methods described by Hendzel and Jamieson (1976).

Round weights or fork lengths for some samples taken in 1985 were estimated when only one of the two measures of fish size was available. Log fork length versus log round weight regressions for each species were calculated using all fish from all other years for each sample area. This regression was used to predict the round weight or fork length of some individual fish taken in 1985.

One-way analysis of variance and one-way analysis of covariance models were used to determine the significance of year-to-year differences in mean mercury concentrations for lake whitefish, walleye and northern pike samples.

Yearly mean mercury concentrations for these three species are reported for each site as three different values: 1) the arithmetic mean; 2) the adjusted mean as derived from an analysis of covariance model (mercury concentration predicted for the grand mean fork length for all years by the "within groups" linear regression of log mercury concentrations versus log fork length (Dixon and Massey 1969)); 3) an adjusted mean calculated separately for each species, year and area sample from the linear regression of log mercury concentration on log fork length at a fixed fork length for each species. The following arbitrary fork lengths were utilized in calculating this latter mean: 350 mm (lake whitefish), 400 mm (walleye), and 550 mm (northern pike).

RESULTS

The results presented in this appendix include mean mercury values for all fish species sampled from a given area in a given year. As noted above, for lake whitefish, walleye and northern pike, mean mercury values are presented as unadjusted means and as means adjusted by two different methods. These values are presented in tables and figures while values for other species are presented in tables only. Results are given in the following tables and figures. See the main body of the report for detailed discussion of these results.

REFERENCES

Dixon, W.J., and F.J. Massey. 1969. Introduction to statistical analysis, Third Edition,. McGraw Hill, New York. 639 p.

Hendzel, M.R., and D.M. Jamieson. 1976. Determination of mercury in fish. Anal. Chem. 48: 926-928.

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Fig. 4. Mean muscle mercury concentrations (adjusted by analysis of covariance) in lake whitefish versus year for five sites on Southern Indian Lake and for Issett Lake.

Fig. 5. Mean muscle mercury concentrations (unadjusted) in walleye versus year for five sites on Southern Indian Lake and for Issett Lake.

Fig. 6. Mean muscle mercury concentrations (adjusted to 400 mm fork length) in walleye versus year for five sites on Southern Indian Lake and for Issett Lake.

Fig. 7. Mean muscle mercury concentrations (adjusted by analysis of covariance) in walleye versus year for five sites on Southern Indian Lake and Issett Lake.

Fig. 8. Mean muscle mercury concentrations (unadjusted) in northern pike versus year for five sites on Southern Indian Lake and for Issett Lake.

Fig. 9. Mean muscle mercury concentrations (adjusted to 550 mm fork length) in northern pike versus year for five sites on Southern Indian Lake and for Issett Lake.

Fig. 10. Mean muscle mercury concentrations (adjusted by analysis of covariance) in northern pike versus year for five sites on Southern Indian Lake and for Issett Lake.

Table 1. Mean muscle mercury concentrations for lake whitefish from Southern Indian Lake and Issett Lake, 1975-1985.

Site	Year	Mean mercury concentration ($\mu g.g^{-1}$)	Range of mercury concentration ($\mu g.g^{-1}$)	Mean mercury concentration standardized to 350mm fork length ($\mu g.g^{-1}$)	Mean mercury concentration adjusted by ANOCOVA ($\mu g.g^{-1}$)	No. of samples	Mean fork length (mm)	Mean round weight (g)
Channel	1975[a]	0.06	0.03-0.12	0.06	0.05	50	332	724
	1978	0.30	0.06-0.60	0.15	0.23	17	423	1438
	1979	0.25	0.04-0.55	0.20	0.24	26	339	804
	1980	0.21	0.02-0.42	0.18	0.16	24	335	711
	1981	0.20	0.04-0.34	0.19	0.18	25	342	752
	1982	0.09	0.03-0.27	0.07	0.07	26	327	549
	1983	0.15	<0.01-0.44	0.08	0.10	34	375	920
	1985[b]	0.15	0.02-0.50	0.07	0.10	44	382	890
Camp 9	1975	0.05	0.03-0.08	0.05	0.05	25	335	721
	1979	0.13	0.06-0.26	0.08	0.10	40	401	1014
	1980	0.13	0.04-0.59	0.09	0.10	28	379	860
	1981	0.10	0.04-0.35	0.09	0.10	24	351	818
	1982	0.09	0.03-0.27	0.06	0.07	25	401	1058
	1983	0.09	0.01-0.21	0.05	0.07	26	385	1007
	1984	0.09	0.02-0.39	0.09	0.09	46	312	472
Region 4	1975	0.05	0.02-0.10	0.05	0.05	25	316	594
	1978	0.22	0.09-0.38	0.21	0.21	16	338	663
	1979	0.10	0.06-0.30	0.09	0.09	68	370	762
	1980	0.14	0.05-0.37	0.13	0.12	27	370	1008
	1981	0.08	0.03-0.32	0.05	0.07	67	358	725
	1982	0.11	0.06-0.21	0.09	0.10	25	390	929
	1983	0.05	<0.01-0.15	0.04	0.04	24	349	746
	1984	0.10	0.01-0.49	0.08	0.08	47	379	870
	1985	0.12	0.04-0.32	0.10	0.10	24	351	704
Region 5	1975	0.07	0.04-0.12	0.07	0.07	25	349	792
	1981	0.26	0.12-0.42	0.26	0.24	25	397	1014
	1982	0.19	0.06-0.36	0.16	0.17	24	388	864
	1984	0.20	0.10-0.46	0.16	0.18	50	408	962
Region 6	1975	0.07	0.03-0.12	0.07	0.07	25	329	728
	1979	0.31	0.05-0.55	0.24	0.24	30	386	1073
	1980	0.20	0.04-0.44	0.17	0.18	20	333	689
	1981	0.14	0.03-0.38	0.14	0.12	26	327	650
	1982	0.11	0.01-0.36	0.09	0.09	37	325	588
	1983	0.15	0.06-0.29	0.14	0.13	28	352	721
	1984	0.16	0.04-0.34	0.12	0.12	38	379	893
	1985	0.09	0.01-0.27	0.08	0.07	50	329	634
Issett Lake	1975	0.15	0.02-0.30	0.14	0.16	24	369	1060
	1978	0.32	0.17-0.40	0.24	0.51	5	310	405
	1982	0.21	0.12-0.36	0.19	0.22	25	384	900
	1983	0.17	0.06-0.28	0.11	0.16	24	406	1137
	1984	0.23	0.05-0.89	0.14	0.20	47	405	1103
	1985	0.25	0.07-0.53	0.10	0.19	50	435	1407

[a] This is a combined sample from region 2 and region 6 (see Fig. 1).

[b] This sample was caught just north of the Channel in region 2 (see Fig. 1).

Table 2. Mean muscle mercury concentrations for walleye from Southern Indian Lake and Issett Lake, 1978-1985.

Site	Year	Mean mercury concentration ($\mu g.g^{-1}$)	Range of mercury concentration ($\mu g.g^{-1}$)	Mean mercury concentration standardized to 400mm fork length ($\mu g.g^{-1}$)	Mean mercury concentration adjusted by ANOCOVA ($\mu g.g^{-1}$)	No. of samples	Mean fork length (mm)	Mean round weight (g)
Channel	1979	0.47	0.25-2.19	0.49	0.47	30	333	476
	1980	0.56	0.29-2.04	0.54	0.51	33	393	756
	1981	0.55	0.36-1.22	0.54	0.51	32	387	711
	1982	0.45	0.23-1.41	0.42	0.41	24	389	669
	1983	0.49	0.25-1.76	0.45	0.44	24	391	708
	1985[a]	0.47	0.26-1.46	0.44	0.42	50	398	771
Camp 9	1979	0.59	0.32-1.80	0.51	0.50	11	406	881
	1980	0.53	0.37-0.76	0.49	0.51	14	388	707
	1981	0.45	0.36-0.51	0.46	0.46	5	362	550
	1982	0.47	0.32-0.65	0.47	0.45	25	406	753
	1983	0.47	0.27-0.63	0.47	0.46	21	390	628
	1984	0.40	0.13-1.22	0.40	0.37	25	365	632
Region 4	1979	0.35	0.30-0.41	0.44	-	3	272	227
	1980	0.57	0.45-0.73	0.61	-	4	380	799
	1981	0.58	0.42-0.79	0.54	-	20	418	982
	1984	0.45	0.31-0.68	0.42	-	25	433	1137
Region 5	1979	0.56	0.35-0.72	0.56	0.56	24	384	725
	1981	0.66	0.39-0.88	0.73	0.72	25	351	569
	1982	0.70	0.37-1.05	0.66	0.62	24	428	913
	1984	0.58	0.25-1.05	0.59	0.55	26	386	695
	1985	0.60	0.33-1.07	0.44	0.50	14	445	1043
Region 6	1978	0.80	0.46-1.20	0.75	0.68	15	435	921
	1979	0.47	0.06-1.14	0.48	0.38	51	365	723
	1980	0.59	0.33-1.03	0.65	0.57	28	376	635
	1981	0.64	0.32-1.94	0.68	0.59	26	382	671
	1982	0.78	0.31-1.66	0.58	0.72	25	355	494
	1983	0.56	0.18-1.38	0.54	0.51	36	400	669
	1984	0.65	0.38-1.49	0.63	0.59	25	385	657
	1985	0.69	0.19-1.66	0.58	0.61	50	369	628
Issett Lake	1978	1.52	1.24-1.95	1.52	1.34	5	389	703
	1982	0.79	0.20-2.52	1.02	0.68	19	347	494
	1983	1.37	0.35-2.71	1.02	0.80	32	446	1123
	1984	0.46	0.07-1.54	0.77	0.70	23	284	374
	1985	1.02	0.33-2.06	0.44	0.45	12	489	1383

[a] This sample was caught just north of the Channel in region 2 (see Fig. 1).

Table 3. Mean muscle mercury concentrations for northern pike from Southern Indian Lake and Issett Lake, 1978-1985.

Site	Year	Mean mercury concentration ($\mu g.g^{-1}$)	Range of mercury concentration ($\mu g.g^{-1}$)	Mean mercury concentration standardized to 550mm fork length ($\mu g.g^{-1}$)	Mean mercury concentration adjusted by ANOCOVA ($\mu g.g^{-1}$)	No. of samples	Mean fork length (mm)	Mean round weight (g)
Channel	1979	0.57	0.29-0.89	0.63	0.64	35	497	917
	1980	0.57	0.05-1.11	0.57	0.49	38	532	1205
	1981	0.64	0.42-1.00	0.62	0.65	25	520	1054
	1982	0.77	0.48-1.09	0.77	0.72	25	542	1024
	1983	0.65	0.34-1.35	0.60	0.54	36	546	1133
	1985[a]	0.75	0.20-1.65	0.71	0.64	44	546	1108
Camp 9	1979	0.58	0.36-1.10	0.57	0.56	35	554	1372
	1980	0.61	0.41-1.01	0.60	0.58	31	566	1326
	1981	0.66	0.43-0.96	0.65	0.65	24	544	1156
	1982	0.68	0.43-0.96	0.65	0.59	25	603	1545
	1983	0.68	0.42-1.07	0.65	0.60	25	600	1509
	1984	0.57	0.16-1.36	0.69	0.60	46	484	824
Region 4	1979	0.49	0.30-1.20	0.50	0.49	54	528	1276
	1980	0.63	0.45-0.91	0.63	0.63	28	539	1355
	1981	0.72	0.19-1.13	0.70	0.68	25	552	1363
	1982	0.63	0.35-0.82	0.61	0.58	24	582	1367
	1984	0.63	0.21-0.98	0.61	0.65	25	500	878
	1985	0.74	0.27-1.32	0.68	0.67	46	559	1274
Region 5	1979	0.62	0.34-1.13	0.65	0.64	40	484	881
	1981	0.68	0.40-1.01	0.69	0.61	25	537	1175
	1982	0.79	0.39-1.23	0.75	0.66	24	571	1227
	1984	0.50	0.19-1.05	0.61	0.48	31	472	752
	1985	0.35	0.29-0.42	-	-	3	542	1100
Region 6	1978	0.77	0.28-1.72	0.61	0.56	15	666	2325
	1979	0.69	0.42-1.21	0.68	0.70	60	530	1209
	1980	0.78	0.42-2.55	0.75	0.75	34	537	1170
	1981	0.89	0.65-1.15	0.88	0.87	25	554	1323
	1982	0.96	0.38-1.54	0.92	0.92	28	552	1155
	1983	0.83	0.15-2.15	0.72	0.71	23	556	1252
	1984	1.18	0.26-2.53	1.09	1.07	36	548	1065
	1985	0.67	0.26-1.11	0.65	0.64	50	541	1168
Issett Lake	1978	0.61	0.37-0.74	0.54	0.70	5	573	1350
	1982	0.90	0.36-1.75	0.76	0.95	26	596	1812
	1983	1.04	0.50-2.28	0.85	0.98	35	658	2450
	1984	1.09	0.25-2.88	0.86	1.09	27	609	1922
	1985	1.12	0.45-2.01	0.73	0.92	50	719	3100

[a] This sample was caught just north of the Channel in region 2 (see Fig. 1).

Table 4. Mean muscle mercury concentrations for cisco, longnose sucker and burbot from Southern Indian Lake and Issett Lake, 1982-1985.

Species	Region	Year	Mean mercury concentration ($\mu g.g^{-1}$)	Range of mercury concentration ($\mu g.g^{-1}$)	No. of samples	Mean fork length (mm)	Mean round weight (g)
Cisco	The Channel	1982	0.16	0.05-0.31	26	261	228
		1983	0.17	0.06-0.37	27	311	478
	Camp 9	1982	0.30	0.13-0.68	11	326	505
		1983	0.15	0.04-0.50	24	294	380
		1984	0.19	0.07-0.63	50	293	359
	Region 4	1982	0.18	0.08-0.75	24	314	439
		1984	0.13	0.06-0.21	21	285	288
	Region 5	1982	0.18	0.07-0.48	24	324	529
		1984	0.19	0.10-0.32	18	242	219
	Region 6	1982	0.18	0.10-0.43	25	259	250
		1983	0.24	0.08-0.58	36	317	533
		1984	0.22	0.06-0.48	32	301	426
	Issett Lake	1982	0.45	0.10-0.91	24	353	754
		1983	0.25	0.12-0.63	24	331	612
		1984	0.35	0.17-0.85	25	348	689
		1985	0.25	0.08-0.62	50	345	655
Longnose sucker	The Channel	1982	0.12	0.07-0.21	24	457	1428
		1983	0.07	0.02-0.15	16	414	1077
	Camp 9	1982	0.08	0.04-0.15	25	425	1306
		1983	0.08	0.04-0.16	25	427	1272
		1984	0.10	0.05-0.24	41	422	1192
	Region 4	1982	0.15	0.07-0.30	24	463	1392
		1984	0.13	0.05-0.47	25	418	1117
	Region 5	1982	0.15	0.07-0.26	24	436	1104
		1984	0.20	0.11-0.27	5	463	1250
	Region 6	1982	0.06	0.04-0.11	7	396	986
		1984	0.13	-	1	414	1520
	Issett Lake	1982	0.13	0.07-0.17	4	458	1625
Burbot	The Channel	1982	0.18	0.14-0.26	4	489	675
	Camp 9	1983	0.18	0.07-0.30	5	494	955
		1984	0.22	0.07-0.60	12	484	900
	Region 4	1982	0.23	0.12-0.39	24	525	938
		1984	0.27	0.18-0.36	2	556	940
	Region 5	1982	0.31	0.12-0.42	13	513	881
		1984	0.48	0.35-0.73	6	578	983
	Region 6	1984	0.19	-	1	495	825

Table 5. Results of statistical analyses used to test for differences in muscle mercury levels over the periods 1975-1985 for whitefish and 1978-1984 (post-impoundment) for whitefish, walleye and northern pike. (* = indicates significance at $p<0.05$ level; ** = indicates significance at $p<0.01$ level; ns = indicates not significant at $p \geq 0.05$ level.

		Years compared (pre- and post-impoundment)		Analysis of variance (Fn,n df)		Analysis of covariance
Species	Site	Pre-	Post-	$\log_{10}$ fork length (x)	$\log_{10}$[Hg] (y)	(Fn,n df)
Lake whitefish (pre- and post-impoundment)	Channel	1975	1978-83,1985	$F_{7,237}$ = 44.5070**	$F_{7,237}$ = 15.7431**	$F_{7,236}$ = 15.9859**
	Camp 9	1975	1979-84	$F_{6,207}$ = 18.6868**	$F_{6,207}$ = 8.9832**	$F_{6,206}$ = 6.8639**
	Region 4	1975	1978-85	$F_{8,306}$ = 4.9973**	$F_{8,306}$ = 24.6254**	$F_{8,305}$ = 23.9071**
	Region 5	1975	1981-82,1984	$F_{3,120}$ = 10.0156**	$F_{3,120}$ = 62.1411**	$F_{3,119}$ = 45.2217**
	Region 6	1975	1979-85	$F_{7,245}$ = 7.8673**	$F_{7,245}$ = 17.4839**	$F_{7,244}$ = 12.5717**
	Issett Lake	1975	1978,1982-85	$F_{5,169}$ = 19.1653**	$F_{5,169}$ = 5.3152**	$F_{5,168}$ = 6.9999**
Lake whitefish (post-impoundment)	Channel	-	1978-83,1985	$F_{6,188}$ = 40.7062**	$F_{6,188}$ = 7.0664**	$F_{6,187}$ = 7.4192**
	Camp 9	-	1979-84	$F_{5,183}$ = 26.3311**	$F_{5,183}$ = 4.0043**	$F_{5,182}$ = 3.7352**
	Region 4	-	1978-85	$F_{7,282}$ = 2.6293*	$F_{7,282}$ = 22.0556**	$F_{7,281}$ = 22.7843**
	Region 5	-	1981-82,1984	$F_{2,96}$ = 2.7988ns	$F_{2,96}$ = 5.0649**	$F_{2,95}$ = 5.1719**
	Region 6	-	1979-85	$F_{6,221}$ = 10.6477**	$F_{6,221}$ = 16.0461**	$F_{6,220}$ = 10.8746**
	Issett Lake	-	1978,1982-85	$F_{4,146}$ = 19.3524**	$F_{4,146}$ = 2.8866*	$F_{4,145}$ = 9.6303**
Walleye (post-impoundment)	Channel	-	1979-83,1985	$F_{5,187}$ = 11.6027**	$F_{5,187}$ = 2.2421ns	$F_{5,186}$ = 1.9863ns
	Camp 9	-	1979-84	$F_{5,95}$ = 3.5876**	$F_{5,95}$ = 3.6243**	$F_{5,94}$ = 2.3988*
	Region 4[1]	-	-	-	-	-
	Region 5	-	1979,1981-82 1984-85	$F_{4,108}$ = 6.4974**	$F_{4,108}$ = 2.9932*	$F_{4,107}$ = 6.4264**
	Region 6	-	1978-85	$F_{7,248}$ = 4.4696**	$F_{7,248}$ = 6.3416**	$F_{7,247}$ = 5.7795**
	Issett Lake	-	1978, 1982-85	$F_{4,86}$ = 22.2904**	$F_{4,86}$ = 13.7118**	$F_{4,85}$ = 5.4279**
Northern pike (post-impoundment)	Channel	-	1979-83,1985	$F_{5,197}$ = 3.0309*	$F_{5,197}$ = 3.7664**	$F_{5,196}$ = 4.6926**
	Camp 9	-	1979-84	$F_{5,179}$ = 12.4541**	$F_{5,179}$ = 3.7961**	$F_{5,178}$ = 1.0464ns
	Region 4	-	1979-82, 1984-85	$F_{5,196}$ = 4.3271**	$F_{5,196}$ = 10.4666**	$F_{5,195}$ = 9.8336**
	Region 5[2]	-	1979, 1981-82 1984	$F_{3,116}$ = 8.6150**	$F_{3,116}$ = 13.3432**	$F_{3,115}$ = 7.6895**
	Region 6	-	1978-85	$F_{7,263}$ = 6.1574**	$F_{7,263}$ = 9.4960**	$F_{7,262}$ = 11.2155**
	Issett Lake	-	1978, 1982-85	$F_{4,138}$ = 6.6561**	$F_{4,138}$ = 3.9483**	$F_{4,137}$ = 3.2638*

---

[1] Sample sizes too small for ANOCOVA.

[2] For 1985, n=3. Therefore 1985 data is excluded from analyses.

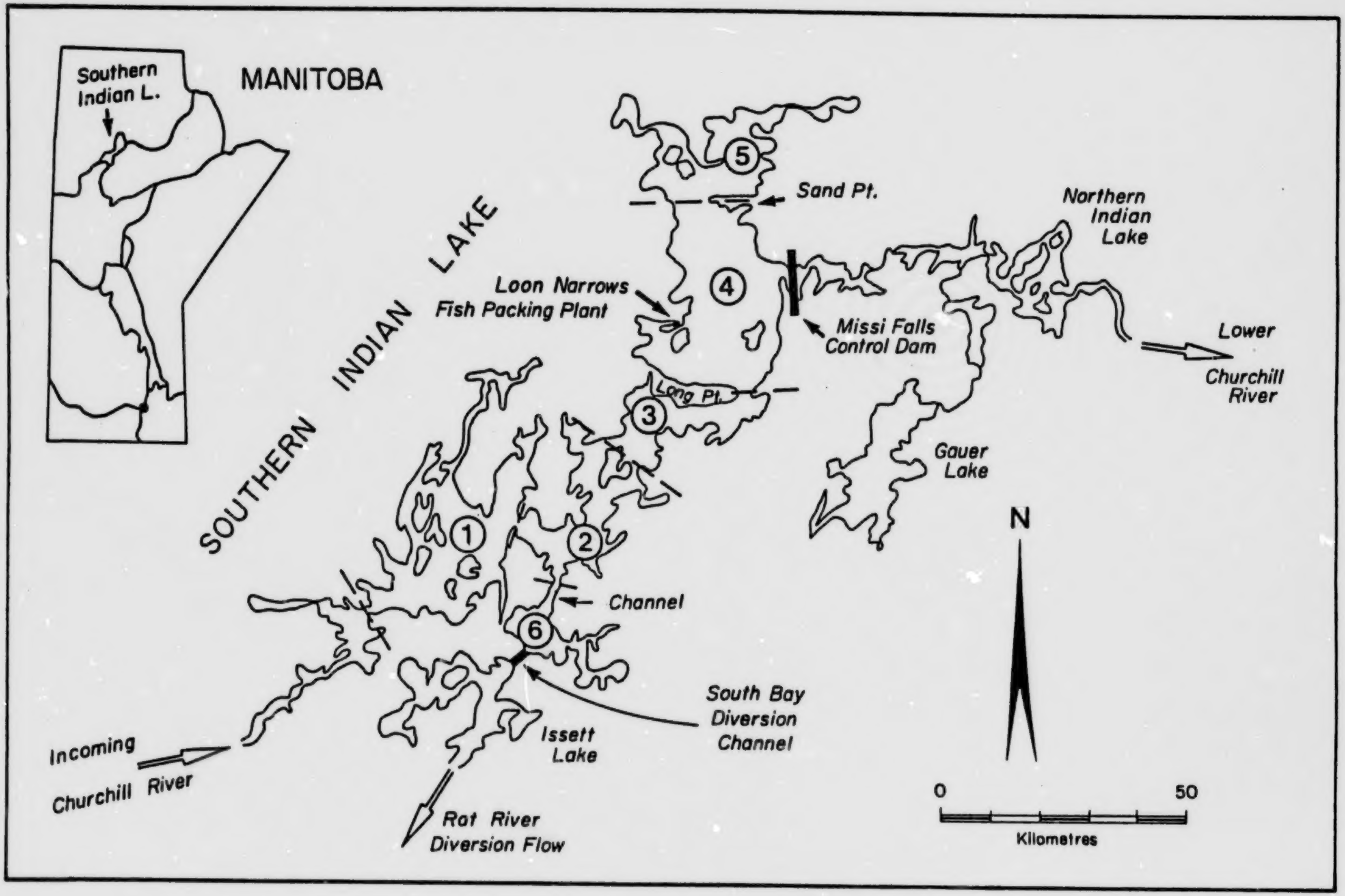
Southern Indian L.
MANITOBA
SOUTHERN INDIAN LAKE
5
Sand Pt.
Northern Indian Lake
Loon Narrows Fish Packing Plant
4
Missi Falls Control Dam
Lower Churchill River
Long Pt.
3
Gauer Lake
1
2
N
Channel
6
South Bay Diversion Channel
Incoming Churchill River
Issett Lake
Rat River Diversion Flow
0
50
Kilometres

Fig. 1 Southern Indian Lake and Issett Lake study sites. Camp 9 = Area 2.

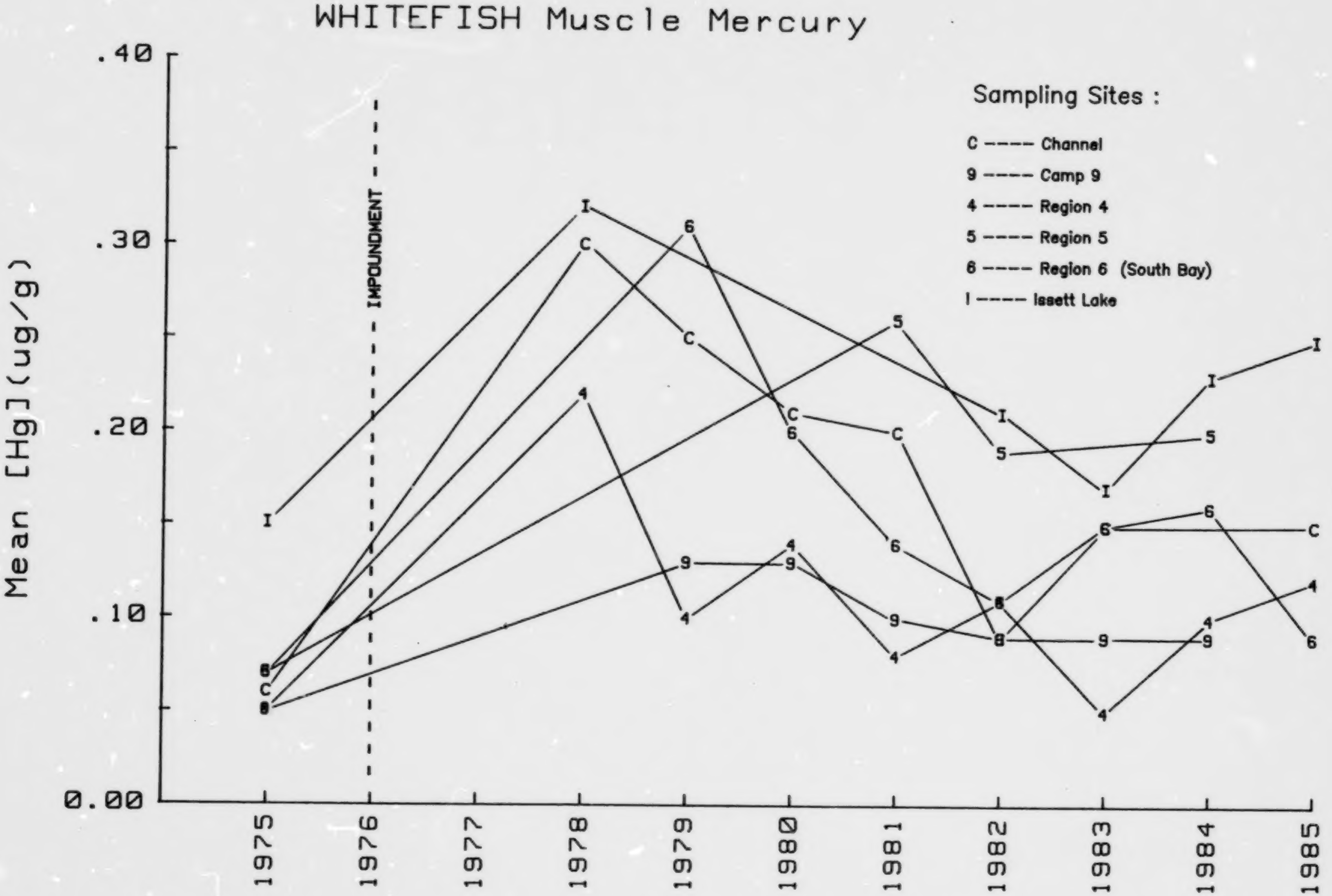

Fig. 2 Mean muscle mercury concentrations (unadjusted) in lake whitefish versus year for five sites on Southern Indian Lake and for Issett Lake.

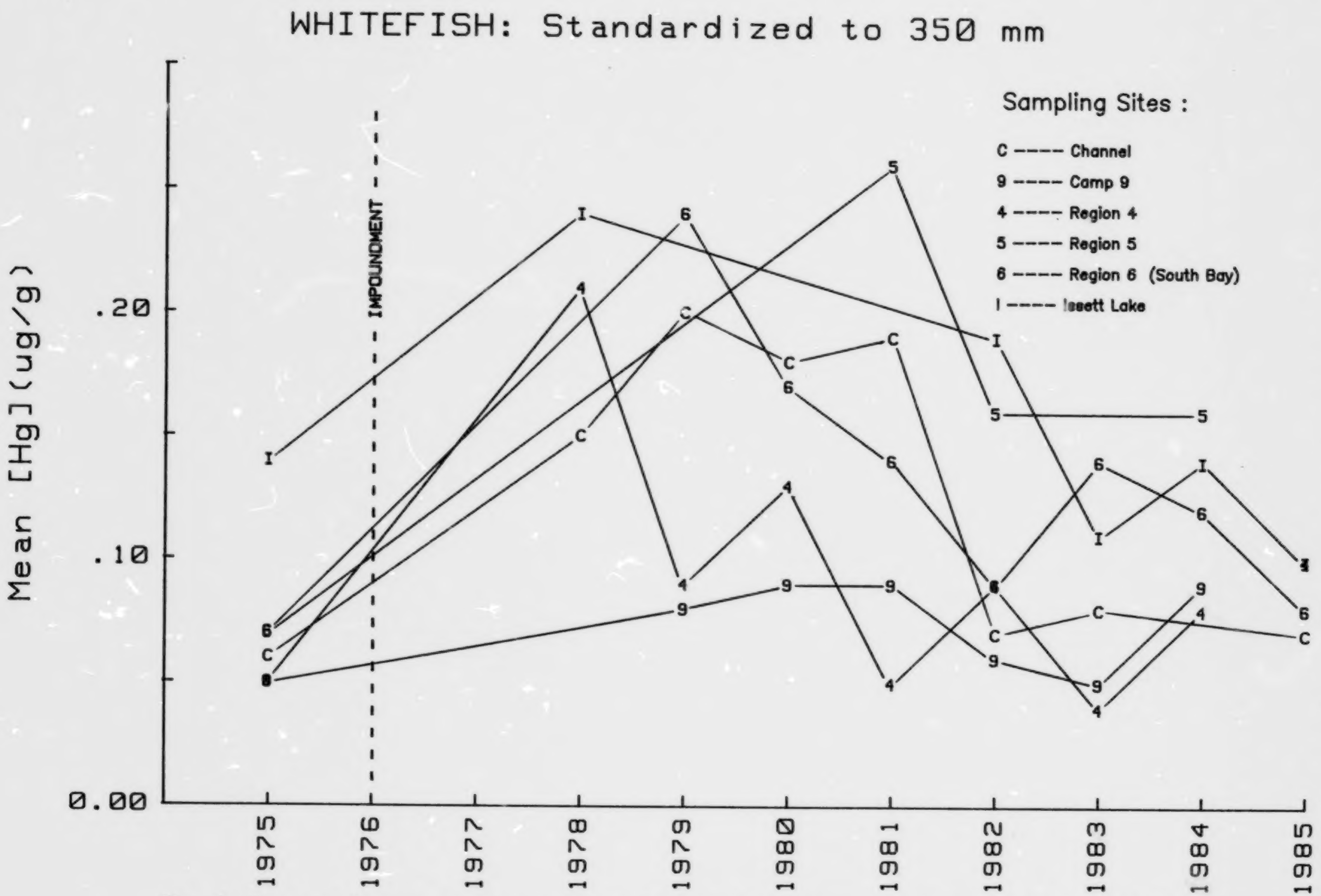

Fig. 3 Mean muscle mercury concentrations (adjusted to 350 mm fork length) in lake whitefish versus year for five sites on Southern Indian Lake and for Issett Lake.

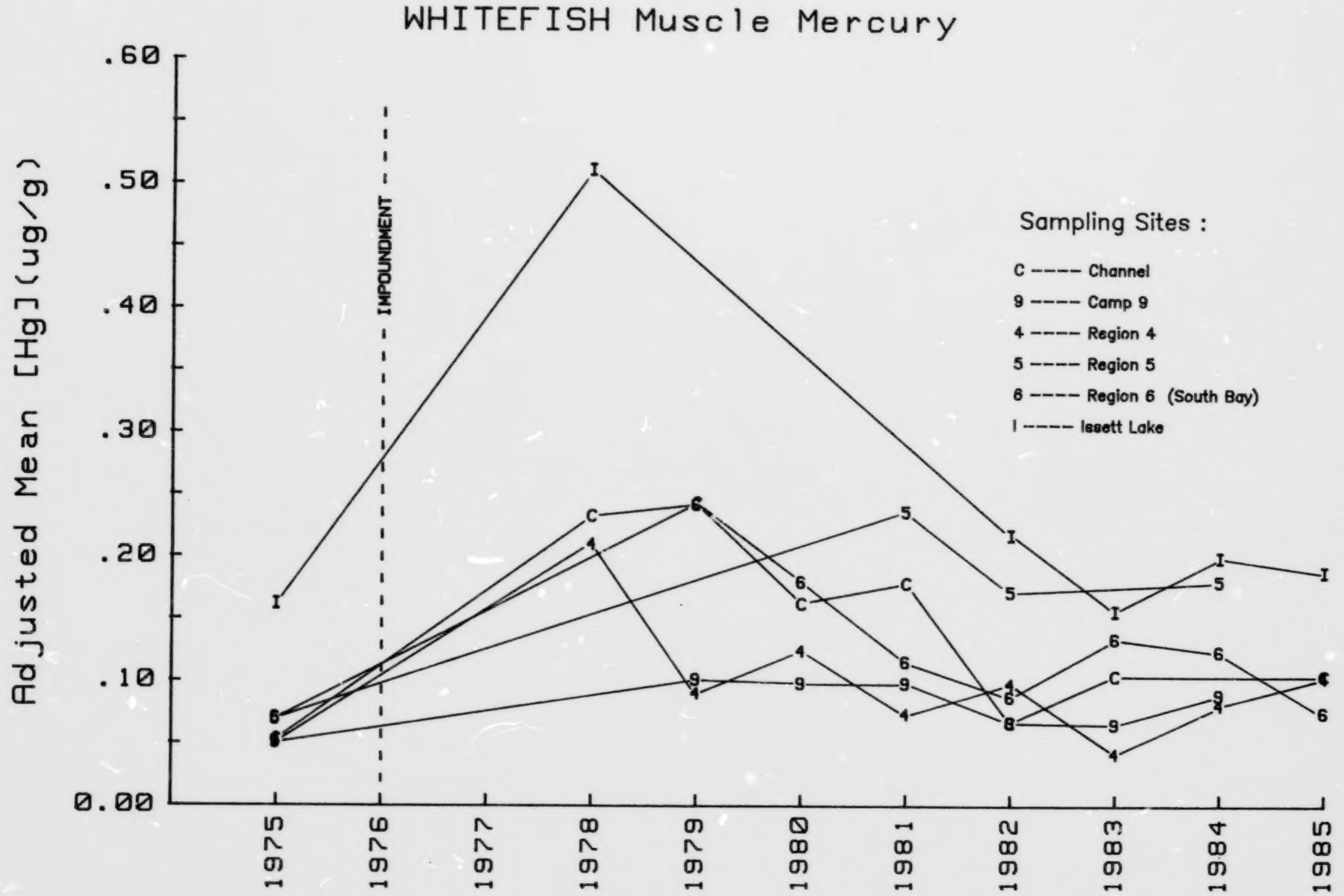

Fig. 4 Mean muscle mercury concentrations (adjusted by analysis of covariance) in lake whitefish versus year for five sites on Southern Indian Lake and for Issett Lake.

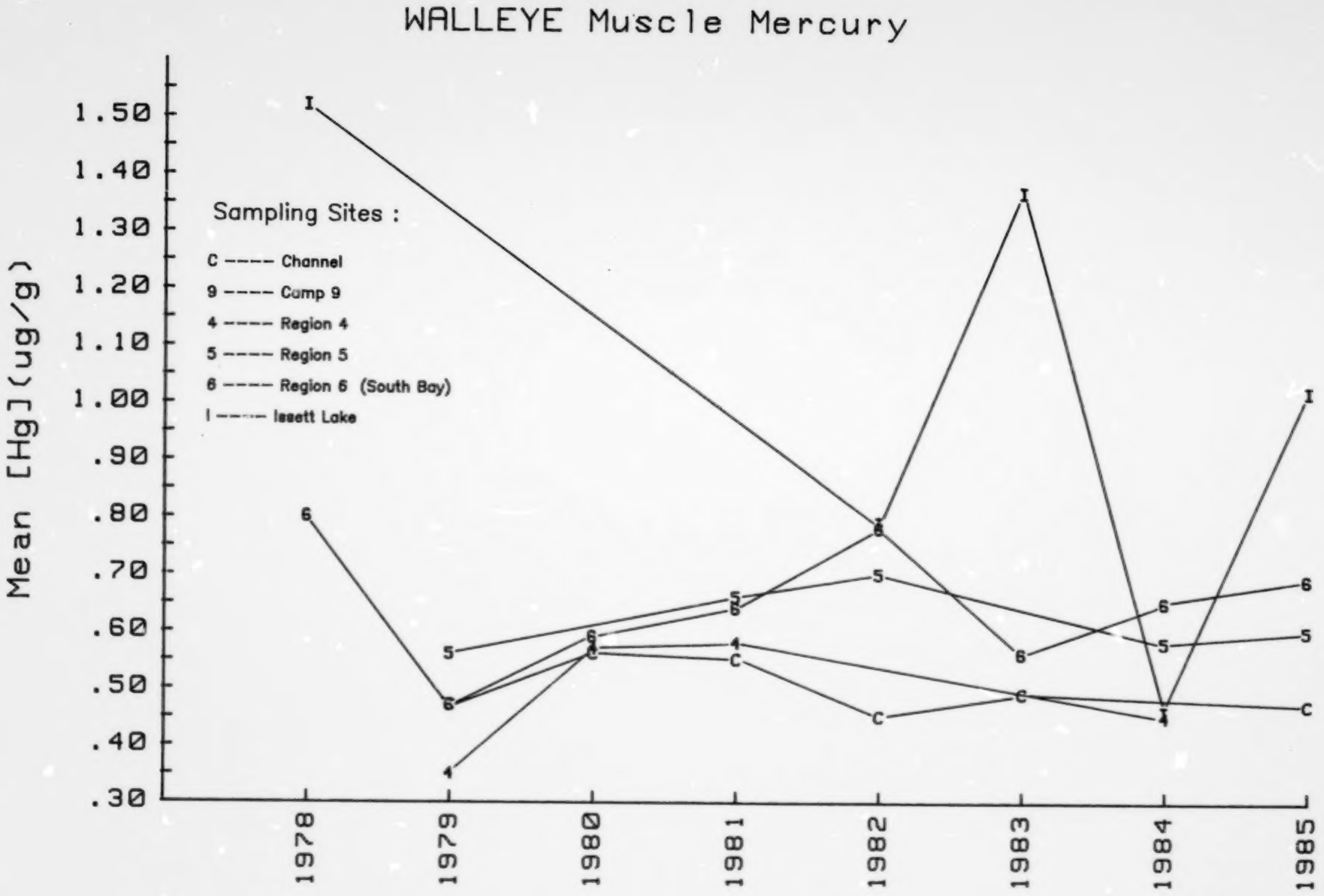

Fig. 5 Mean muscle mercury concentrations (unadjusted) in walleye versus year for five sites on Southern Indian Lake and for Issett Lake.

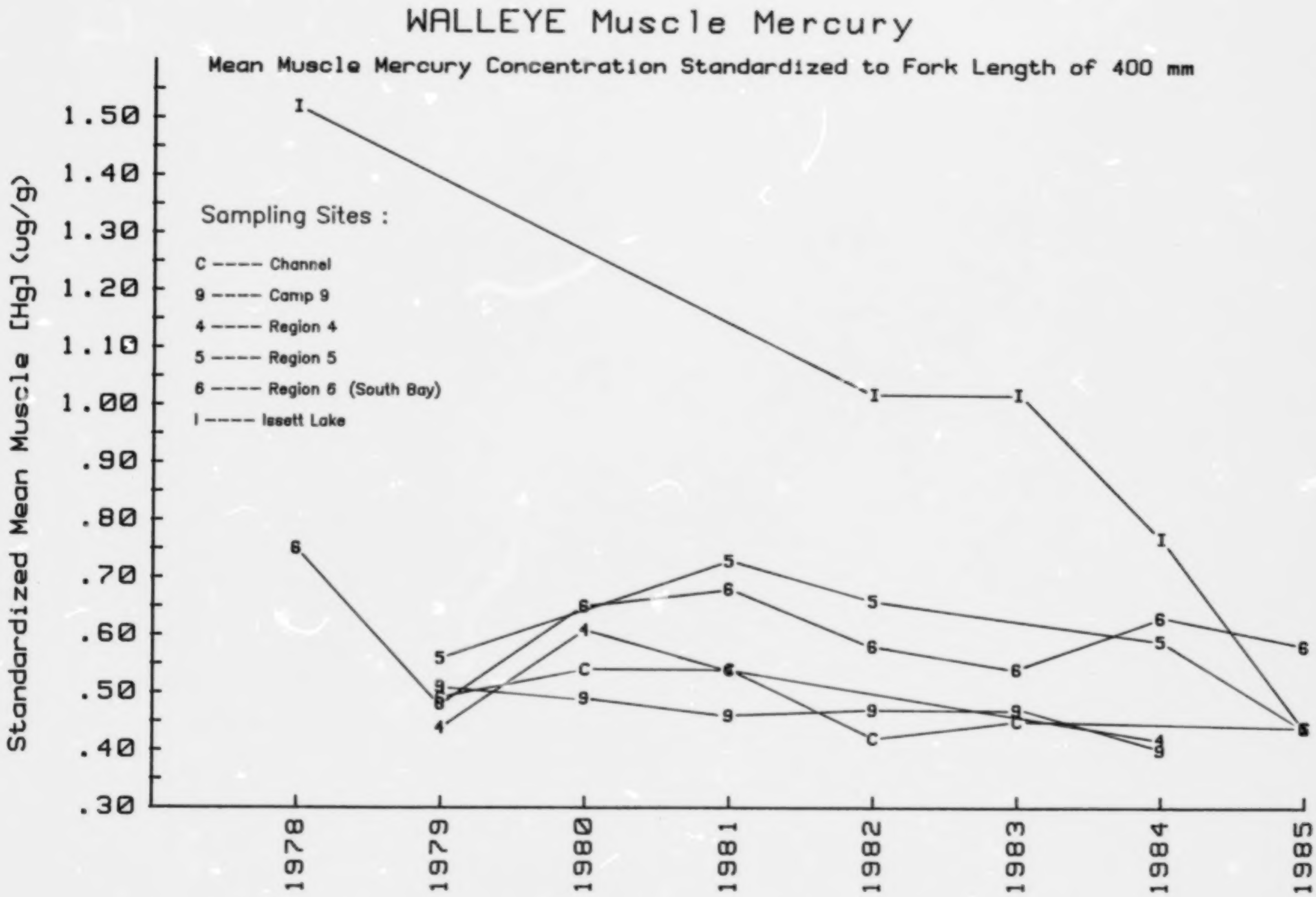

Fig. 6 Mean muscle mercury concentrations (adjusted to 400 mm fork length) in walleye versus year for five sites on Southern Indian Lake and for Issett Lake.

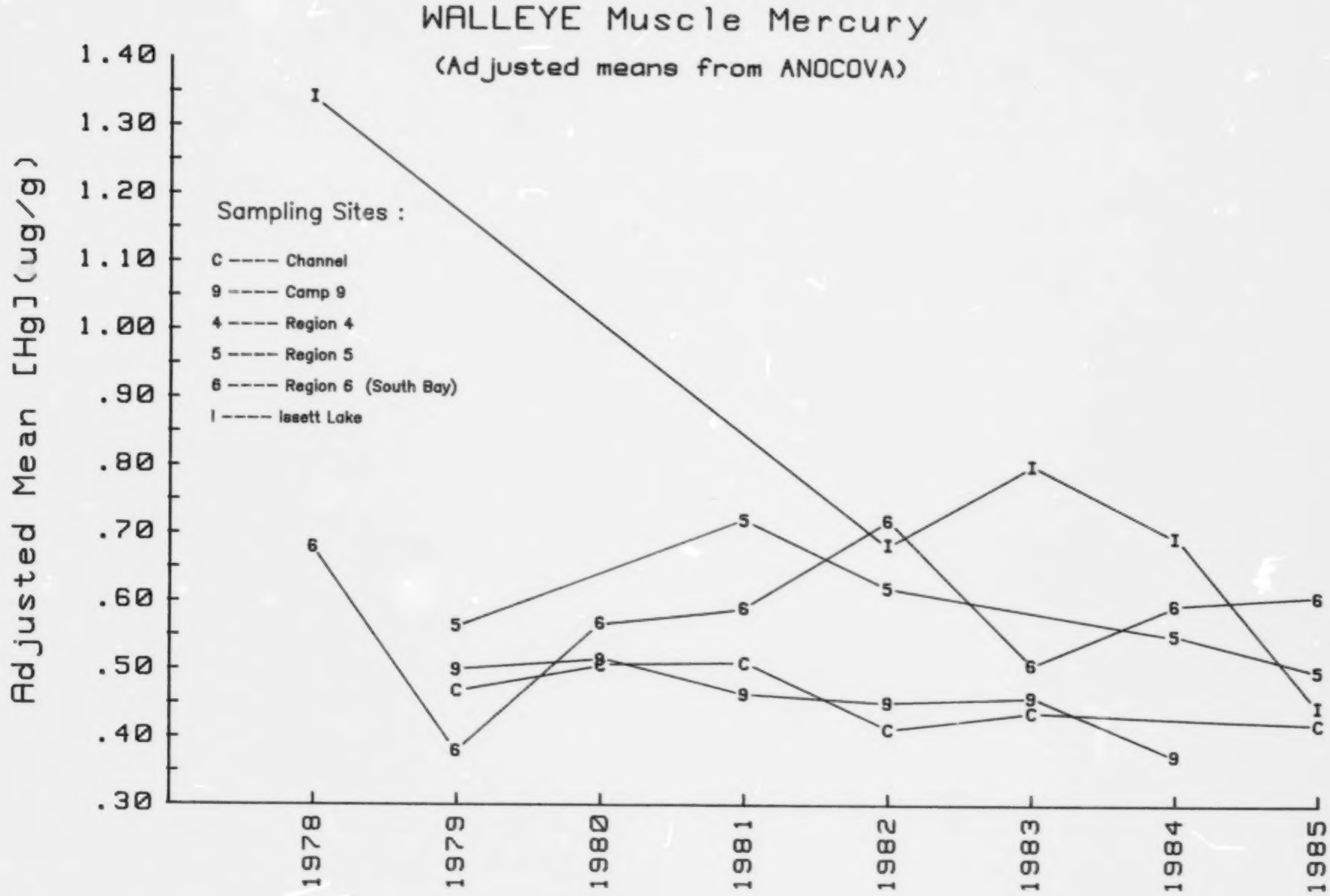

Fig. 7 Mean muscle mercury concentrations (adjusted by analysis of covariance) in walley versus year for five sites on Southern Indian Lake and Issett Lake.

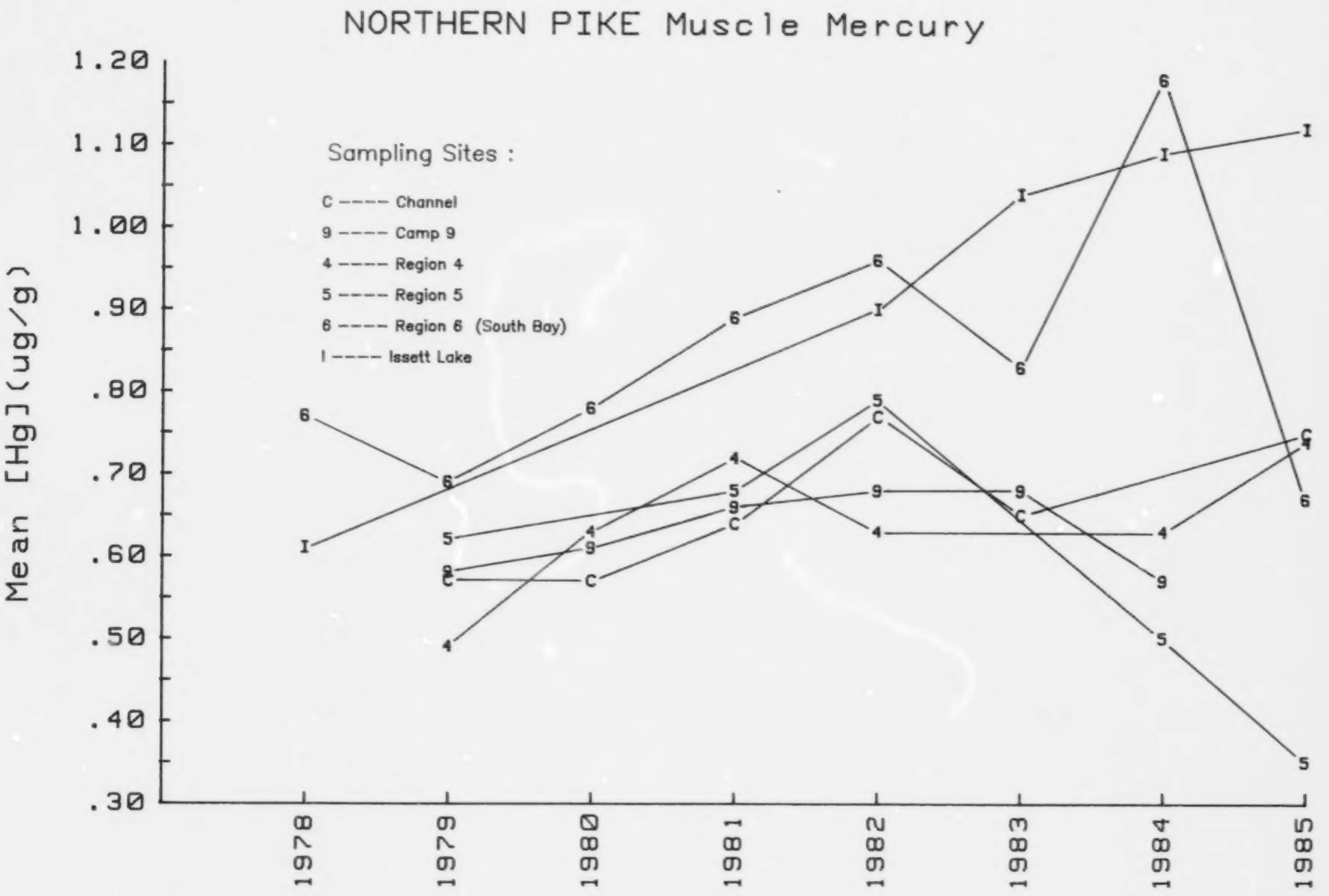

Fig.8 Mean muscle mercury concentrations (unadjusted) in northern pike versus year for five sites on Southern Indian Lake and for Issett Lake.

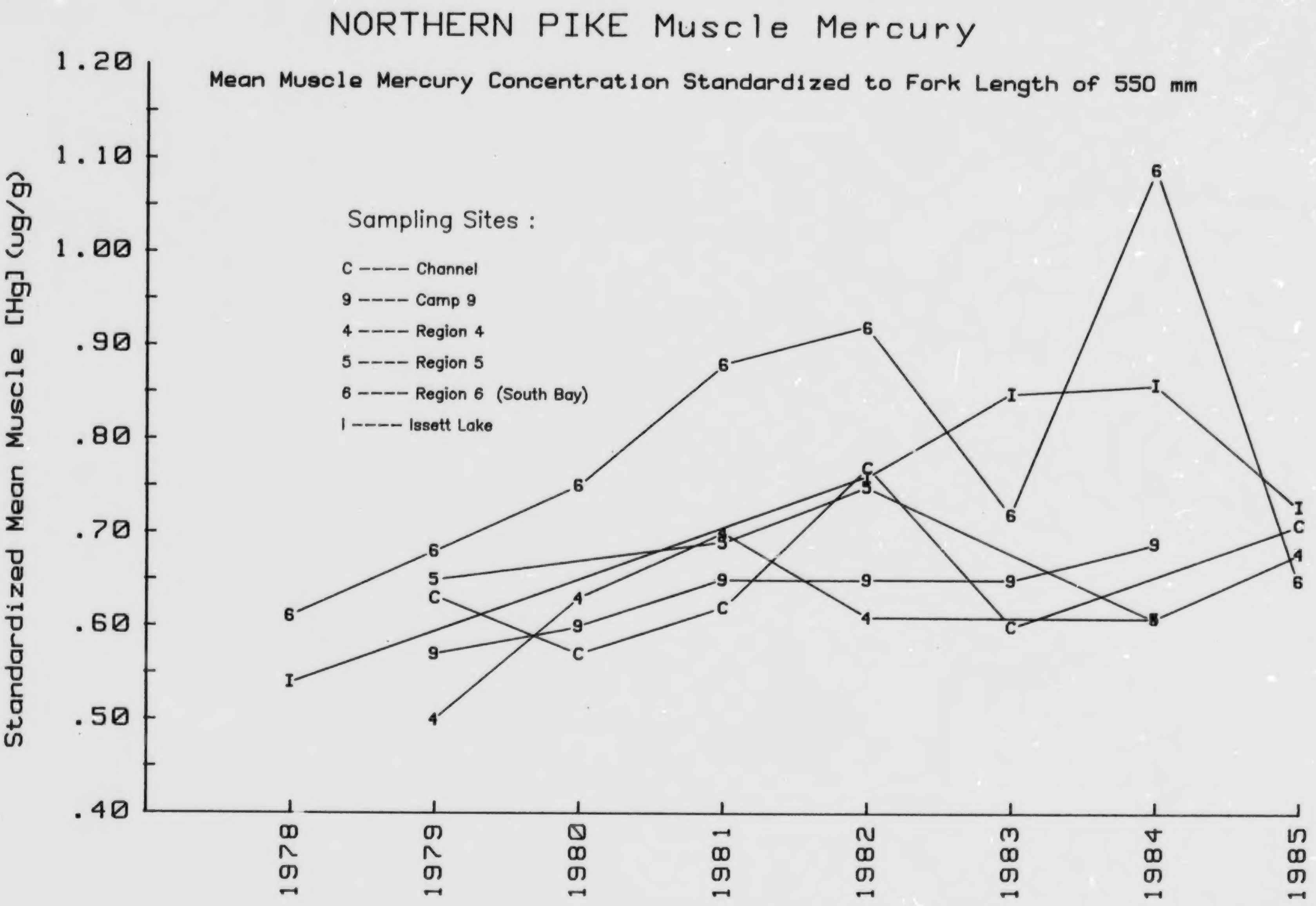

Fig. 9 Mean muscle mercury concentrations (adjusted to 550 mm fork length) in northern pike versus year for five sites on Southern Indian Lake and for Issett Lake.

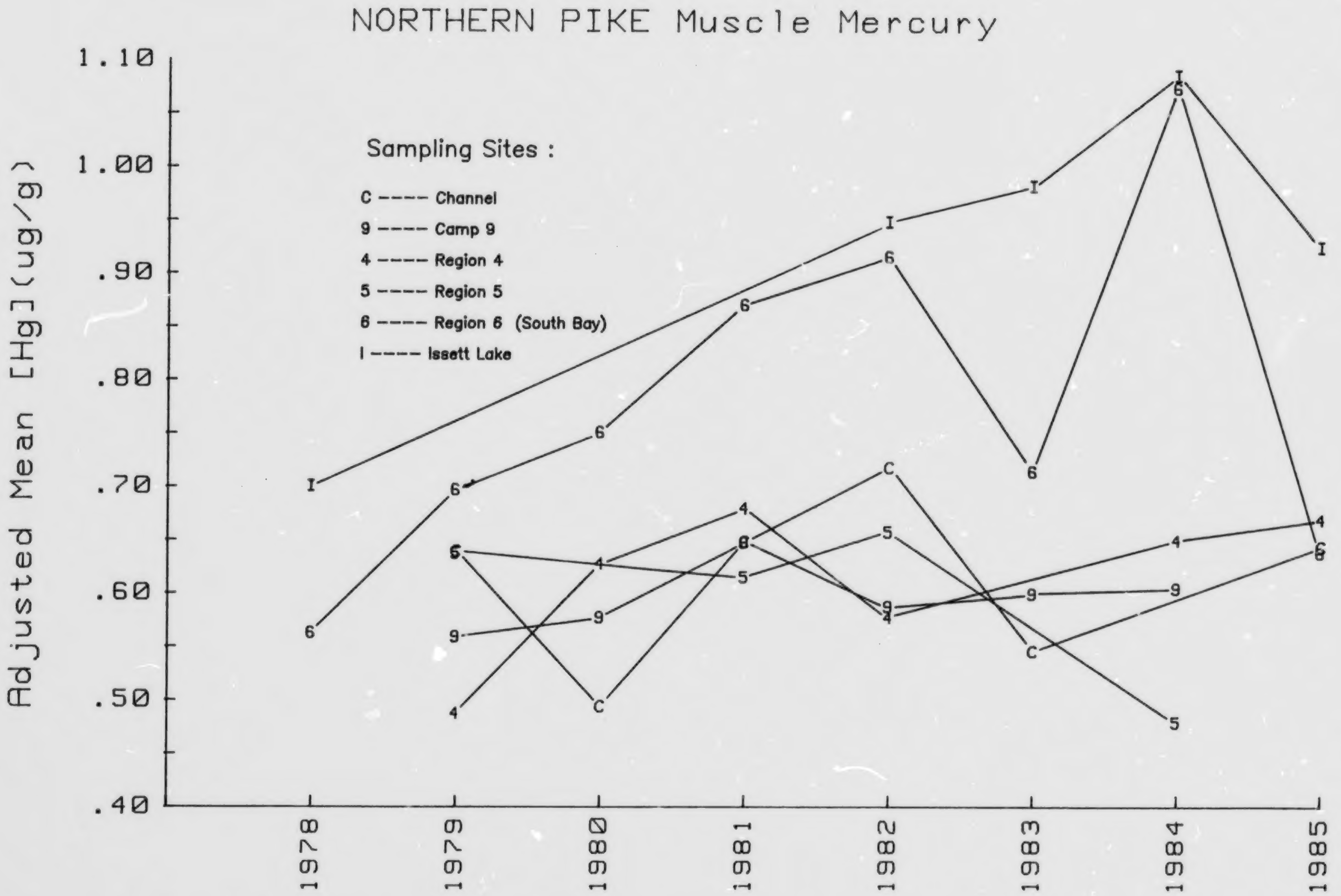

Fig. 10 Mean muscle mercury concentrations (adjusted by analysis of covariance) in northern pike versus year for five sites on Southern Indian Lake and for Issett Lake.

Volume

4

Canada Manitoba

15. Mercury in Mink, Otter, and Small Mammals from the Churchill River Diversion, Manitoba
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16. Mercury Data from Inspected Commercial Shipments of Fish from the Churchill River Diversion Area in Manitoba
Inspection Branch, Fisheries and Oceans Canada

17. Total Mercury Concentrations in Large Fishes from lakes on the Churchill River Diversion and Nelson River
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Medical Services Branch, Health and Welfare Canada

Canada - Manitoba Agreement on the Study and Monitoring of Mercury in the Churchill River Diversion

Technical Appendices to the Summary Report Volume 4

MERCURY IN MINK, OTTER, AND SMALL MAMMALS FROM THE CHURCHILL RIVER DIVERSION, MANITOBA

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Terrestrial Standards and
Studies
Manitoba Department of
Environment and Workplace
Safety and Health

SPECIAL NOTE

This report was prepared for the Canada-Manitoba Agreement on the Study and Monitoring of Mercury in the Churchill River Diversion. Any conclusions, recommendations, or opinions expressed herein are those of the author and are not necessarily the same as those of the Steering Committee for this Agreement.

EXECUTIVE SUMMARY

The objective of this Project, no. 3.1 under the Canada-Manitoba Mercury Agreement, was to determine whether there is accumulation of mercury in food chain higher than fish, to determine its geographical extent, and to determine if there was a discernible upward or downward trend in the mercury levels.

Mink, otter, and small mammals were collected from different locations within the Agreement area and from control areas. Their tissues were analyzed for mercury content. Mink and otter were monitored because trapping for fur provides income for the area residents, and because these animals are suitable indicators of environmental contamination by mercury. Small mammals are prey of both the aquatic and the terrestrial predators

The results of this study indicate that accumulation of mercury in the food chain in the Agreement area does occur. The findings are consistent with the hypothesis that the mercury is mobilized as a result of flooding.

Mercury concentrations in the tissues of mink and otter in the Agreement area were higher than in most areas of southern Manitoba. A small percentage of the aquatic furbearers accumulated near-toxic concentrations of mercury in their tissues.

Records collected by the Manitoba Department of Natural Resources show no apparent relation between the trapping success over the last four seasons and the mercury concentrations. There is no indication that the current mercury levels have any adverse effects on the mink and otter populations. Mercury concentrations in mink have decreased over the past two years, but no conclusion can be reached regarding long-term trends.

Most of the conclusions drawn from the results are based on the analyses of mink samples. However, otter and small mammal data are in support of the mink results.

In areas with elevated mercury concentrations where fish are caught for human consumption, mammal monitoring cannot replace the sampling of fish. For routine monitoring of other areas, however, mink can serve as well as, or better than fish. As a rule of thumb, assuming a ten-to-one mink:fish ratio of mercury concentrations, fish monitoring would become necessary when mean mercury concentrations in mink liver exceeded 5 ug/g freeze dried weight or approximately 2.0 ug/g fresh weight. Lower values could be used for added margin of safety. Toxic effects of mercury are manifested in mink and man at similar concentrations, but humans do not consume as much fish, therefore this type of monitoring would provide ample warning of any impending risk to people.

It is recommended that monitoring of aquatic carnivores of the area be continued, both to establish the long-term trends and to provide necessary up-to-date information. In addition, a more extensive project should be undertaken to evaluate the potential of small mammal surveys for mercury monitoring.

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INTRODUCTION

This paper is the final report on the results of Project 3.1, Mercury in Fauna, undertaken as part of the Canada - Manitoba Agreement on the Study and Monitoring of Mercury in the Churchill River Diversion (further referred to as "the Agreement"). The objective of Project 3.1 was to determine "whether there is accumulation [of mercury] further along the food chain, and if such accumulation is verified, to determine its geographical extent."

Between 1974 and 1978, a hydroelectric development project in northern Manitoba, Canada, diverted most of the flow of the Churchill River into the Nelson River, via the Rat and Burntwood Rivers, to augment the power production of existing plants on the Nelson. Necessary engineering controls include two large dams, one at the outlet of Southern Indian Lake on the Churchill River, the other at Notigi Lake on the Burntwood River. As a result, water level in Southern Indian Lake was raised by 3 m, and extensive flooding occurred both on the lake and along the Diversion route.

Post-impoundment mercury (Hg) concentrations in fish from the Diversion system increased to such level that commercial fishing in lakes of the Diversion was discontinued, and some fish from the Southern Indian Lake had to be marketed in countries that had less stringent guidelines for acceptable mercury concentrations in fish, e.g. 1.0 ppm. Health and environmental concerns led to the signing of the Agreement in 1983.

Carnivores were monitored for two principal reasons. Firstly, trapping for fur provides income for the area residents, which makes it important to know whether the animal populations are in danger. Secondly, these animals are suitable indicators -- or sentinels -- of environmental contamination, because of their position near to top of the food chain. Previous studies from both Manitoba (Kucera 1983) and elsewhere (Halbrook 1978, Wren et al. 1980) documented that carnivorous aquatic mammals accumulate mercury in their tissues in excess of the concentrations in their prey. In Manitoba, mink (*Mustela vison*) and river otter (*Lutra canadensis*) tissue concentrations of Hg were in direct proportion to those found in fish from the same drainage areas (Kucera 1983). The similarity of toxic levels of mercury in mink and man (Skerfving 1972 a,b) further emphasizes the importance of this biomonitoring project.

Post-impoundment increases of mercury in fish from the Churchill River Diversion system have been well documented by Bodaly and Hecky (1979) and Bodaly et al. (1984). Their generally accepted hypothesis is that elevated mercury in the aquatic system is due to bacterial methylation of naturally occurring mercury in flooded soils. It follows that elevated mercury concentrations could be expected in tissues of fish-eating mammals from the flooded areas. The carnivores may also respond to year-to-year changes in the levels of mercury in the environment. Accordingly, this study compares mercury concentrations in mammals from different regions of the Agreement area with those of control areas. Year-to-year changes were examined to determine if there was a discernible upward or downward trend in the mercury levels.

STUDY SPECIES

Carnivores

Mink and otter are the two aquatic carnivores at the top of the freshwater food chain in the study area. Fish are their staple food, supplemented by other prey in the water such as clams, crayfish, and frogs. Home ranges of mink encompass on the average 1.85 km of a stream for the females and 2.6 km for the males. Males may also hunt inland (Linscombe et al. 1982). Sometimes a third or more of the diet of the mink consists of small mammals. Although the otter has a much larger home range than the mink, which also includes inland area, fish is the major component of its diet (Wise et al. 1981). Crayfish are another important food of the otter (Toweill and Tabor 1982).

Small Mammals

The deer mouse (*Peromyscus maniculatus*) and the red-backed vole (*Clethrionomys gapperi*) are the most abundant small mammal species that occur in the study area. Both are important prey for the terrestrial predators as well as for mink. They provide a potential link between the aquatic and the terrestrial food chain. In addition, because these mammals have small home ranges (generally less than 2 ha), results can be related directly to the site of capture.

STUDY AREAS

Within the Agreement area, carnivore samples were obtained from trappers operating in the following four Registered Trapline Sections (RTS) (Figures 1 and 2):

(i) Southern Indian Lake (SIL),
(ii) Nelson House (NEL),
(iii) Split Lake (SLK), and
(iv) Pikwitonei (PIK).

Samples from southern Manitoba were included for comparison. Pooled data from samples that originated from the Red River, Assiniboine River, and Dauphin River drainage basins (Figure 3) served as a control. An additional set of 11 mink and two otter from the Winnipeg River drainage (mercury contaminated) was evaluated separately.

Small-mammal samples were collected near

(i) Notigi Lake,
(ii) Threepoint Lake, and
(iii) Odei River (unflooded control).

MATERIALS AND METHODS

From fall 1981 to spring 1985, 403 mink and 55 otter carcasses were obtained from trappers. Of these, 224 mink and 53 otter carcasses were collected from the four northern areas, the remaining samples came from the southern parts of Manitoba (Tables 1 and 2).

The collection, tissue sampling, and chemical analysis followed a previously established protocol (Kucera 1983). Skinned carcasses were obtained from licensed or registered trappers, and processed in the laboratories of Manitoba Department of Environment and Workplace Safety and Health. Samples of brain, liver, and kidney tissues were analyzed for total mercury by the method of Hendzel and Jamieson (1976). Liver and kidney tissues are the principal body organs that concentrate mercury. Brain tissue was analyzed because the central nervous system is the primary target of ingested organically bound mercury, and most affected by mercury toxicity (Nordberg and Skerfving 1972). A commercial laboratory (Matson's, Milltown, Montana) determined the ages, using the tooth cementum annuli method.

The small mammal survey was undertaken during the summer of 1984. Ninety-three small rodents (four species) were collected from the three locations. In each case they were trapped on two parallel traplines, one along the shoreline, and the other 100 - 200 m inland, beyond the range of any spray from the water. The skins (including fur) of the rodents were analyzed for mercury by the same method as the carnivore tissues.

Standard statistical tests (Steel and Torrie 1960) were performed using programs available in SAS and APL systems on the Manitoba Data Services mainframe computer. Since the concentrations of Hg in tissues did not follow a normal distribution, nonparametric tests using median rather than mean values were usually employed to compare Hg concentrations between areas, years, etc. In general, however, the same trends were observed when comparing means. To compare data from different areas, analysis of covariance was performed where appropriate, using means adjusted for sex and year.

RESULTS

Mink

Table 3 presents the age/sex distribution of the mink catch. Eighty percent or more mink were younger than 2 years, and over 60 percent were young of the year.

Four-way factorial analysis of variance on ranked data was used to examine variations in tissue Hg concentrations on the basis of age, sex, location, and year. The mercury concentrations varied significantly with location, year and sex (F-test, $p<0.01$). There were no significant interactions.

Appendix A contains detailed descriptive statistics regarding mercury concentrations in brain, liver, and kidney tissues in different years and different areas. Mean, median, mode, maximum and minimum values, and representative percentiles are included; data from animals of both sexes and all ages were pooled.

The mercury concentrations were highest in SIL and lowest in PIK (Figure 4). Though higher in SIL, median (as well as mean) concentrations in samples from SIL and NEL were not significantly different (Figure 5). The SLK samples had significantly lower mercury concentrations than SIL and NEL samples, while the concentrations in animals from PIK were even lower than the southern Manitoba controls (Tables 4-6). This same pattern existed when means were compared and regardless of whether the means were adjusted for sex or year or both (Table 7).

Female mink accumulated significantly more mercury in brain, liver, and kidney tissues than the males did (F test; $p<0.01$). Mercury concentrations in these three tissues were significantly correlated. Spearman Correlation Coefficients (p<0.01) were r=0.90 for brain vs. liver, 0.93 for brain vs. kidney, and 0.91 for liver vs. kidney) (Table 8).

In a previous study (Kucera 1983), ratios of brain Hg to liver+kidney Hg were used to assess the relative exposure of animals from different areas to organically bound mercury. Regressions of brain vs. liver+kidney Hg show that brains of mink from northern areas accumulated a relatively higher proportion of Hg. Because neural tissues are the primary target of organically bound Hg, this indicates that mink from the North may ingest more organic mercury than those from southern Manitoba (Table 9). Comparisons between individual areas (Table 10) suggest that mink from NEL, SIL and PIK are more exposed to organically bound Hg than other mink. A note of caution is in order regarding the use of the ratio of the Hg concentration in brain versus that in other tissues because the method has not yet been confirmed by laboratory studies (Kucera 1983). Since generally over 90 percent of the total Hg in fish muscle is in the organic form (Sherbin 1979), it has not been verified if the higher ratio reflects an increased intake of organic mercury per se, or whether it is simply because of a higher total mercury insult. The ingested mercury is partly demethylated in mammalian body (Jernelov 1976). O'Connor and Nielsen (1981) estimated the methyl mercury content of mink and otter livers as about half of the total. However, most of the available literature data on mercury toxicity in wild mammals present total concentrations (Kucera 1983).

Although the analysis of variance did not show a significant overall effect of age on the Hg concentrations, individual comparisons revealed that animals older than four years had accumulated significantly greater concentrations of mercury. The lack of significance when the effect of age was tested may be a reflection of the low number (< 2%) of older animals in the sample (Table 3).

Tables 4, 5, and 6 present the data to show changes in Hg concentrations from year to year. Data from SLK are available for three seasons, and from SIL and NEL for two seasons. Mercury concentrations increased from 1983 to 1984 at SLK, although the difference was not statistically significant. The concentrations in NEL and SLK decreased significantly from 1984 to 1985, while those in SIL also appeared to decrease, but the change was not statistically significant. Data from southern Manitoba are available for 1983 and 1984; they indicate no significant difference in Hg levels between the two years. Compared to the Agreement area, the Hg concentrations in southern Manitoba were consistently lower in both 1983 and 1984. Appendix A includes concentrations in a sample of 11 mink from the Winnipeg River drainage in 1982. These values were similar to those found in northern Manitoba. This supports the theory of increased Hg accumulation with increased exposure because the Winnipeg River is known to be contaminated with mercury.

Otter

Tables 11-13 summarize the data on mercury in otter The trends appear to be similar to those observed in mink, although the differences are less definite and not always statistically significant. The positive correlation between age and concentration of Hg in liver ($r = 0.46$) and kidney tissue ($r = 0.47$) was significant ($p<0.05$). However, the small number of otters does not allow statistically meaningful comparisons between years or areas. The age/sex distribution of the otter samples is presented in Table 14.

Unlike mink, correlations between sex and Hg accumulation were not significant in otters (Table 8). Samples from all animals were pooled for calculation of the descriptive statistics shown in Appendix B.

Small Mammals

Of the 93 small mammals, only the deer mice were caught in sufficient numbers (59) to allow statistical comparisons. Mercury concentrations in their hides were highest at Notigi Lake, and lowest at Odei River (control). Notigi results were significantly higher than those from Threepoint or Odei (U test, $p<0.05$). Furthermore, in all three areas the animals caught along the water's edge had higher mercury concentrations than animals caught inland The differences were not significant when tested for each area individually (U test, $p<0.08$) but were significant ($p<0.01$) when all water-edge and all inland samples were pooled. (Table 15-A).

Concentrations of Hg in hides of red-backed voles do not show significant differences between areas (Table 15-B). Although the number of samples was too low (21) for proper statistical comparison, when the samples from all three locations were pooled, the difference between water-edge and inland animals also approached statistical significance (U test, $p<0.056$).

DISCUSSION

Most of the conclusions drawn from the results are based on the analyses of mink samples. However, otter and small mammal data do not contradict any of the mink results, and generally support the same conclusions.

Mercury concentrations in mink samples have decreased over the past two seasons. When compared to the results of the 1979/1980 pilot study (Kucera 1983), it appears that the concentrations in mink from the Nelson House RTS increased after 1980, but fell between 1984 and 1985. In the absence of a continuous time series, the pattern is unclear.

The concentrations in mink tissues were, for the most part, within the nontoxic range, but higher than those reported from other parts of Manitoba (Kucera 1983). One liver sample from SIL and one liver sample from SLK exceeded the levels considered to be toxic (Figure 5). However, because other tissues of the same individuals had normal concentrations, the samples are being reanalyzed to see if the result was in error. Toxic concentrations (recalculated to ug/g freeze-dried weight) which are reported in the literature are
36, 59, and 54 for mink,
57, 81, and 91 for otter;
(Wobeser and Swift 1976, Wobeser et al.1976, O'Connor and Nielsen 1981, Wren 1985).

Mercury concentrations at which mink begin to exhibit some symptoms of poisoning are 21, 44, and 42 ug/g for brain, liver, and kidney, respectively (Wobeser et al. 1976).

Interpretations based on the small mammal data are inhibited by the small sample size which was a product of time, budget, and laboratory schedule constraints. However, the data do complement the mink results. The Diversion and the flooding are again implicated as the source of the mercury accumulated by mice. The higher concentrations of Hg in *Peromyscus* compared to *Clethrionomys* point to bioaccumulation; diet of *Peromyscus* includes a large complement of invertebrates (Wolff et al. 1985) while *Clethrionomys* is a more herbivorous species.

Higher mercury levels in mice from the water edge may contribute to the explanation of why female mink concentrate more Hg than males; the wider-ranging male may eat relatively more small mammals from farther inland (Banfield 1974). The more sedentary female's diet may contain more fish, shells, and crayfish as well as small mammals with higher mercury levels. Another explanation which has been suggested is that the Hg concentrations in females may be higher because of higher metabolic rates combined with the smaller size (Wren, personal communication). Magos et al.(1981), who found more Hg in female than in male rats treated with methyl mercury, attributed their finding to greater loss of growth and body weight in the females. Although O'Connor and Nielsen (1981) reported higher concentrations in female mink and otter from Eastern United States, in Manitoba Kucera (1983) and this study found the opposite.

Effects on furbearer populations

Records collected by the Manitoba Department of Natural Resources indicate that after an unexplained reproductive failure during 1982/83 the populations of mink across entire Western Canada remained at a lower level for two years, followed by some improvement in 1984/85. In the 1985/86 trapping season the Manitoba mink population appears to have returned to normal (R.Stardom, Natural Resources, personal communication).

Fur production records from the four northern areas are presented in Table 16. The pattern bears out the above information. The trapping success over the last four seasons shows no apparent relation to the mercury concentrations. There is no indication that the current mercury levels have any adverse effects of on the mink and otter populations.

O'Connor and Nielsen (1981) predicted that mercury poisoning would occur in mink and otter populations when fish would regularly contain 2.0 ppm (wet weight) of methyl mercury. As the mercury concentrations in fish from the Agreement area did not reach those levels (see below), there is little reason to expect any detrimental effects either on furbearer populations or on the fur harvest in the area.

Comparison with data on fish

An extensive amount of information regarding mercury in fish has been accumulated by both federal and provincial investigators. Table 17 summarizes available data on one common predatory fish species, the northern pike (R.A.Bodaly, Fisheries and Oceans, J.O'Connor and D.Green, Natural Resources, personal communication). The ranking of mercury concentrations in mammals from different areas does not correspond to that obtained by analyses of fish samples. Most of the fish data show that the SIL concentrations were lower than those from Notigi and some of the other Diversion lakes. Year-to-year changes in fish and mammal mercury levels did not correspond, either. However, in both fish and mammals the mercury levels were lowest for Split Lake, and elevated concentrations were found in northern Manitoba compared to southern Manitoba.

The mercury concentrations in mammal samples decreased progressively with distance downstream of Southern Indian Lake along the Diversion route. While the flooding of the lake may have been the primary cause, and the decrease downstream due to dilution factors, such explanation would not take into account the (much more) extensive flooding that occurred elsewhere along the Diversion. For example, in the Notigi reservoir the percentage increase in the flooded area was greater than that in the Southern Indian Lake itself (Bodaly et al. 1984). The suggestion of a "dilution effect" was probably a sampling artifact related to the distribution of sampling areas (Figures 1 and 2). While practically all of the SIL samples were from locations that were subjected to the flooding, the NEL Registered Trapline Section included many areas unaffected by the flooding.

Yearly changes in mercury concentrations in mink correspond only in a very general way to similar data in fish (Table 17). That is, there was some increase after 1979, and a decrease in latter years. However, the year-to-year variability in the fish data does not follow a consistent pattern either, and for many species long-term trends are still not completely clear (Bodaly, Green, personal communication). There are several possible reasons for the year-to-year fluctuations. Changes in water regime and erosion patterns caused by climatic changes may in turn affect the Hg release and the methylation/demethylation processes. A superimposed natural rhythm, or a delayed response by the predator are also possible factors.

Use of mammals for monitoring

As discussed above, the RTS include areas that are not in direct contact with the Diversion proper. In addition, some of the traplines might be located away from waters directly affected by the flooding. This, though inevitable, is a drawback when direct comparisons with fish data are attempted. Results of mink and otter tissue analyses are best interpreted for a whole drainage area, unless exact locations are known. Small mammal monitoring, should it prove feasible and reliable, is free of such problems. However, the present data on small mammals can not provide a proper basis for evaluation because the sample size was too small.

In areas with elevated mercury concentrations where fish are caught for human consumption, mammal monitoring cannot obviate the sampling of fish. For routine monitoring of other areas, however, mink can serve as well as, or better than fish. The tissue concentrations of mercury in mink are approximately ten times higher than those in fish from the same drainage systems (Kucera 1983). Therefore mink would be the better *sentinel* species (*sensu* Newman 1975), i.e., the one that would provide the first warning of increasing mercury levels in a system. As a rule of thumb, assuming a ten-to-one mink:fish ratio of mercury concentrations, fish monitoring would become necessary when mean mercury concentrations in mink liver exceeded 5 ug/g freeze dried weight or approximately 2.0 ug/g fresh weight. Lower values could be used for added margin of safety. Toxic effects of mercury are manifested in mink and man at similar concentrations, but humans do not consume as much fish, therefore this type of monitoring would provide ample warning of any impending risk to people.

CONCLUSIONS

Results of this study indicate that accumulation of mercury in the food chain in the Agreement area does occur. The findings are consistent with the hypothesis that the mercury is mobilized as a result of flooding.

Mercury concentrations in the tissues of mink and otter in the Agreement area were higher than in most areas of southern Manitoba.

A small percentage of the aquatic furbearers accumulated near-toxic concentrations of mercury in their tissues. However, there does not appear to be any detrimental effect on their populations or on the fur harvest in the area.

Mercury concentrations in mink have decreased over the past two years, but without a continuous time-series of data no conclusion can be reached regarding long-term trends.

RECOMMENDATIONS

Monitoring of aquatic carnivores of the area should be continued, both to establish the long-term trends and to provide necessary up-to-date information.

Periodic monitoring of mink from other drainage basins would provide an inexpensive system of early warning regarding potential mercury contamination from any source.

A more extensive research project should be undertaken to confirm or refute the data obtained in the small mammal survey and to assess the potential of such studies for mercury monitoring.

ACKNOWLEDGMENTS

Mr. D. C.M.Elliott provided invaluable assistance in setting up the carcass collections and helping with the preliminary data analysis. Miss L.Uin completed the final data entry, and helped with statistical analysis and presentation of the data. Terrestrial Standards and Studies Section of the Manitoba Department of Environment, Workplace Safety and Health provided funds for all of the collections of data outside the Agreement area, and for collecting during years not covered by the Agreement funding. Natural Resources personnel were instrumental in arranging contacts with the trappers and transport of the carcasses. Mr. O.Hudson was especially helpful. Comments on the manuscript by Dr.R.A. Bodaly and Dr.C.D. Wren are gratefully acknowledged.

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TABLES

Table 1. Number of mink carcasses collected from different areas of Manitoba from 1981/82 to 1984/85.

AREA	YEAR				AREA TOTAL
	1981/82	1982/83	1983/84	1984/85	
Southern Indian Lake	0	0	18	9	27
Nelson House	0	0	12	38	50
Split Lake	0	13	14	43	70
Pikwitonei	0	0	0	77	77
Winnipeg River	11	0	0	0	11
AGREEMENT AREA[1] TOTAL	0	13	44	90	147
SOUTHERN MANITOBA TOTAL	0	103	65	0	168
MANITOBA TOTAL	11	116	109	167	403

[1] Agreement area = Southern Indian Lake, Nelson House, and Split Lake Registered Trapline Sections.

Table 2. Number of otter carcasses collected from different areas of Manitoba from 1982/83 to 1984/85.

AREA	YEAR			AREA TOTAL
	1982/83	1983/84	1984/85	
Southern Indian Lake	0	3	1	4
Nelson House	0	2	0	2
Split Lake	1	21	6	28
Pikwitonei	0	0	19	19
Mantario Lake[1]	2	0	0	2
AGREEMENT AREA[2] TOTAL	1	26	7	34
MANITOBA TOTAL	3	26	26	55

[1] Winnipeg River drainage basin

[2] Agreement area = Southern Indian Lake, Nelson House, and Split Lake Registered Trapline Sections.

Table 3. Age distribution of mink collected from the Agreement area[1] and Pikwitonei Registered Trapline Section (pooled) and from Southern Manitoba regions from 1981/82 to 1984/1985.

AREA	YEAR	SEX	AGE (years)							TOTAL	
			.5	1.5	2.5	3.5	4.5	5.5	6.5	%	N
Agreement Area and Pikwitonei (pooled)	1982/83	M	80	0	20	0	0	0	0	100	5
		F	63	25	0	13	0	0	0	100	8
	1983/84	M	83	10	3	3	0	0	0	100	30
		F	70	30	0	0	0	0	0	100	10
	1984/85	M	80	14	2	2	0	0	1	100	97
		F	86	5	5	5	0	0	0	100	63
Southern Manitoba	1981/82[2]	M	50	33	0	17	0	0	0	100	6
		F	80	0	20	0	0	0	0	100	5
	1982/83	M	68	21	7	0	4	0	0	100	57
		F	74	14	2	7	2	0	0	100	43
	1983/84	M	59	18	18	5	0	0	0	100	22
		F	69	8	15	0	0	0	8	100	13

[1] Agreement area = South Indian Lake, Nelson House, and Split Lake Registered Trapline Sections.

[2] 1981/82 southern Manitoba sample from Winnipeg River area only.

NOTE: Animals of unknown age or sex excluded from table.

Table 4. Mercury levels in brain tissue of Manitoba mink, µg/g freeze-dried tissue [2]. Median (range).

A. Individual Areas

AREA	YEAR			OVERALL YEARS
	1982/83	1983/84	1984/85	
Southern Indian Lake	--	4.38^{a}_{a} (0.29-19.7)	2.49^{a}_{a} (0.79-16.2)	4.00^{a}
Nelson House	--	3.62^{a}_{a} (1.93-12.0)	1.83^{a}_{a} (0.20-22.4)	2.18^{a}
Split Lake	2.29^{a}_{a} (0.34-6.45)	2.33^{a}_{a} (0.96-7.75)	0.93^{b}_{b} (0.11-4.48)	1.19^{b}
Pikwitonei	--	--	0.72^{c} (0.13-6.86)	0.72^{c}

B. Area Totals

AREA	YEAR			OVERALL YEARS
	1982/83	1983/84	1984/85	
Agreement Area [1]	2.29^{a}	3.62^{a}_{a}	1.47_{b}	1.93^{a}
Southern Manitoba	0.80^{b}_{a} (0.1-4.35)	0.93^{b}_{a} (0.13-4.02)	--	0.86^{b}

[1] Agreement area = Southern Indian Lake, Nelson House, and Split Lake R.T.S.

[2] To calculate micrograms per gram wet weight, multiply by 0.33

NOTE: Within each column, medians with different superscripts are significantly different from each other (Kruskal-Wallis test, $P < 0.05$).

Within each row, medians with different subscripts are significantly different from each other (Kruskal-Wallis test, $P < 0.05$).

Table 5. Mercury levels in liver tissue of Manitoba mink, µg/g freeze-dried tissue [2]. Median (range).

A. Individual Areas

AREA	YEAR			OVERALL YEARS
	1982/83	1983/84	1984/85	
Southern Indian Lake	--	$10.90^{a,b}_{a}$ (2.09-74.7)	7.18^{a}_{a} (3.97-43.0)	9.92^{a}
Nelson House	--	15.90^{a}_{a} (5.14-42.4)	7.85^{a}_{b} (0.52-55.6)	8.61^{a}
Split Lake	8.68^{a}_{a} (1.40-18.5)	8.10^{b}_{a} (2.32-26.3)	3.63^{b}_{b} (0.37-62.1)	4.57^{b}
Pikwitonei	--	--	2.60^{b} (0.30-21.7)	2.60^{b}

B. Area Totals

AREA	YEAR			OVERALL YEARS
	1982/83	1983/84	1984/85	
Agreement Area [1]	8.68^{a}	12.05^{a}_{a}	5.48_{b}	6.62^{a}
Southern Manitoba	2.73^{b}_{a} (0.29-29.4)	3.71^{b}_{a} (0.36-19.5)	--	3.13^{b}

[1] Agreement area = Southern Indian Lake, Nelson House, and Split Lake R.T.S.

[2] To calculate micrograms per gram wet weight, multiply by 0.41

NOTE: Within each column, medians with different superscripts are significantly different from each other (Kruskal-Wallis test, $P < 0.05$).

Within each row, medians with different subscripts are significantly different from each other (Kruskal-Wallis test, $P < 0.05$).

Table 6. Mercury levels in kidney tissue of Manitoba mink, µg/g freeze-dried tissue [2]. Median (range).

A. Individual Areas

AREA	YEAR			OVERALL YEARS
	1982/83	1983/84	1984/85	
Southern Indian Lake	--	12.25^{a}_{a} (1.69-48.1)	6.13^{a}_{a} (2.41-29.5)	10.2^{a}
Nelson House	--	9.49^{a}_{a} (4.46-23.6)	5.06^{a}_{a} (0.59-39.0)	6.04^{a}
Split Lake	5.49^{a} (1.21-12.0)	5.24^{a}_{a} (1.72-24.5)	2.95^{b}_{b} (0.36-12.1)	3.79^{b}
Pikwitonei	--	--	2.19^{b} (0.16-13.2)	2.19^{c}

B. Area Totals

AREA	YEAR			OVERALL YEARS
	1982/83	1983/84	1984/85	
Agreement Area [1]	5.45^{a}	9.37^{a}_{a}	4.25_{b}	4.98^{a}
Southern Manitoba	2.44^{b}_{a} (0.39-13.2)	3.18^{b}_{a} (0.56-13.7)	--	2.82^{b}

[1] Agreement area = Southern Indian Lake, Nelson House, and Split Lake R.T.S.

[2] To calculate micrograms per gram wet weight, multiply by 0.43

NOTE: Within each column, medians with different superscripts are significantly different from each other (Kruskal-Wallis test, $P < 0.05$).

Within each row, medians with different subscripts are significantly different from each other (Kruskal-Wallis test, $P < 0.05$).

Table 7. Mean Hg levels in brain, liver, and kidney tissue of mink (µg/g freeze-dried tissue) after adjustment for year and sex through analysis of covariance.

1. Brain Hg means:

		ADJUSTED FOR		
	Unadjusted	Year	Year, Sex	Sex
South Indian Lake	5.01^a	4.82^a	4.96^a	5.15^a
Nelson House	3.90^a	4.11^a	4.08^a	3.88^a
Split Lake	1.78^b	1.70^b	1.66^b	1.78^b

2. Liver Hg Means:

		ADJUSTED FOR		
	Unadjusted	Year	Year, Sex	Sex
South Indian Lake	15.51^a	14.89^a	15.96^a	16.55^a
Nelson House	13.11^a	13.79^a	13.56^a	12.94^a
Split Lake	7.20^b	6.93^b	6.68^b	6.91^b

3. Kidney Hg means:

		ADJUSTED FOR		
	Unadjusted	Year	Year, Sex	Sex
South Indian Lake	12.38a	11.98^a	12.38^a	12.78^a
Nelson House	9.41^a	9.85^a	9.77^a	9.34^b
Split Lake	4.67^b	4.50^b	4.40^b	4.56^c

NOTE: Within columns, means with different superscripts are significantly different from each other (SNK Test, $P < 0.05$).

Table 8. Spearman rank correlation coefficients of mercury levels in brain (Hg-B), liver (Hg-L), and kidney (Hg-K) tissues with age, sex, and weight of mink and otter (all years and areas pooled).

A. Mink

	Hg-B	Hg-L	Hg-K	age	sex	weight
Hg-B	1.00*	0.90*	0.93*	-0.04	-0.13*	-0.24*
Hg-L	0.90*	1.00*	0.91*	-0.01	-0.18*	-0.26*
Hg-K	0.93*	0.91*	1.00*	-0.00	-0.09*	-0.20*

* Significant correlation, $p < 0.01$.

B. Otter

	Hg-B	Hg-L	Hg-K	age	sex	weight
Hg-B	1.00*	0.89*	0.87*	-0.26	-0.05	-0.17
Hg-L	0.89*	1.00*	0.87*	-0.46*	-0.01	-0.26
Hg-K	0.87*	0.87*	1.00*	-0.48*	-0.12	-0.29

* Significant correlation, $p < 0.01$.

Table 9. Slopes of regression of brain Hg on liver plus kidney Hg for mink and otter from various areas of Manitoba, 1982/1983 to 1984/1985

Area	Mink	n	Otter	n
Agreement Area [1]	0.16^{a}	145	0.16^{a}	20
Southern Manitoba	0.13^{b}	167	--	--
Pikwitonei	0.16^{a}	76	0.08^{a}	19

[1] Agreement area = Southern Indian Lake, Nelson House and Split Lake Registered Trapline Sections

NOTE: Values with different superscripts are significantly different from each other (F-test, $p < 0.05$).

Table 10. Slopes of regression of brain Hg on liver plus kidney Hg for mink from various areas of Manitoba, 1982/1983 to 1984/1985

Area	n	Slope
Southern Indian Lake	27	0.16[a]
Nelson House	50	0.18[a]
Split Lake	68	0.09[b]
Pikwitonei	76	0.16[a]
Southern Manitoba	167	0.13[c]

NOTE: Values with different superscripts are significantly different from each other (F-test, $p < 0.05$).

Table 11. **Mercury levels in brain tissue of otters, µg/g freeze-dried tissue [2]. Median (range).**

AREA	YEAR			OVERALL YEARS
	1982/83	1983/84	1984/85	
Southern Indian Lake	--	4.33^{a}_{a} (2.09-24.6)	4.15^{a}_{a} (N = 1)	4.24a
Nelson House	--	$4.08^{a,b}$ (2.56-5.6)	--	$4.08^{a,b}$
Split Lake	0.50_{a} (N = 1)	1.13^{b}_{a} (0.66-8.41)	0.80^{a}_{a} (0.65-1.6)	$0.10^{b,c}$
Pikwitonei	--	--	0.80^{b} (0.35-2.64)	0.80^{c}
Agreement Area[1]	0.50	1.27	0.78	1.18

[1] Agreement area = Southern Indian Lake, Nelson House, and Split Lake R.T.S.

[2] To calculate micrograms per gram wet weight, multiply by 0.33

NOTE: Within each column, medians with different superscripts are significantly different from each other (Kruskal-Wallis test, $p < 0.05$).

Within each row, medians with different subscripts are significantly different from each other (Kruskal-Wallis test, $p < 0.05$).

Table 12. Mercury levels in liver tissue of otters, µg/g freeze-dried tissue [2]. Median (range).

AREA	YEAR			OVERALL YEARS
	1982/83	1983/84	1984/85	
Southern Indian Lake	--	17.10^{a}_{a} (5.2-17.3)	11.60^{a}_{a} (N = 1)	14.35a
Nelson House	--	11.67^{a} (8.33-15.0)	--	11.67^{a}
Split Lake	3.03^{a} (N = 1)	12.10^{a}_{a} (3.28-43.0)	4.32^{a}_{a} (2.01-17.8)	4.87^{a}
Pikwitonei	--	--	3.51^{a} (1.26-14.5)	3.51^{b}
Agreement Area[1]	3.03	13.55	5.00	9.04

[1] Agreement area = Southern Indian Lake, Nelson House, and Split Lake R.T.S.

[2] To calculate micrograms per gram wet weight, multiply by 0.41

NOTE: Within each column, medians with different superscripts are significantly different from each other (Kruskal-Wallis test, $p < 0.05$).

Within each row, medians with different subscripts are significantly different from each other (Kruskal-Wallis test, $p < 0.05$).

Table 13. Mercury levels in kidney tissue of otters, µg/g freeze-dried tissue [2]. Median (range).

AREA	YEAR			OVERALL YEARS
	1982/83	1983/84	1984/85	
Southern Indian Lake	--	15.30^{a}_{a} (6.13-31.2)	21.60^{a}_{a} ((N = 1)	18.45a
Nelson House	--	18.15^{a} (12.4-23.9)	--	18.15^{a}
Split Lake	3.04^{a}_{a} (N = 1)	7.37^{a}_{a} (2.84-35.7)	3.20^{a}_{a} (2.18-11.1)	$4.48^{a,b}$
Pikwitonei	--	--	3.90^{a} (1.28-11.1)	3.90^{b}
Agreement Area [1]	3.04	12.10	3.64	8.25

[1] Agreement Area = Southern Indian Lake, Nelson House, and Split Lake R.T.S.

[2] To calculate micrograms per gram wet weight, multiply by 0.43

NOTE: Within each column, medians with different superscripts are significantly different from each other (Kruskal-Wallis test, $p < 0.05$).

Within each row, medians with different subscripts are significantly different from each other (Kruskal-Wallis test, $p < 0.05$).

Table 14. Age distribution of otters collected from the Agreement area[1] and the Pikwitonei Registered Trapline Section (pooled) during 1983/1984 and 1984/1985.

			Age (years)												Total	
Area	Year	Sex	0.5	1.5	2.5	3.5	4.5	5.5	6.5	7.5	8.5	9.5	10.5	11.5	%	N
Agreement area	1983/84	M	0	33	33	0	0	0	33	0	0	0	0	0	100	3
		F	38	25	25	0	12	0	0	0	0	0	0	0	100	8
Agreement area+Pikwitonei	1984/85	M	67	13	7	0	0	0	0	0	0	7	0	7	100	15
		F	60	20	0	0	10	0	10	0	0	0	0	0	100	10

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[1] Agreement area = South Indian Lake, Nelson House, and Split Lake Registered Trapline Sections

NOTE: Animals of unknown age or sex are excluded from the table

Table 15 Mercury concentrations in hides of small mannals caught along water-edge and inland traplines at three different locations within the Agreement area.

A. DEER MICE (Peromyscus maniculatus)

LOCATION	WATER EDGE			INLAND		
	N	Mean[1]	Median[2]	N	Mean[1]	Median[2]
Notigi Lake	19	5.40^{a}	3.42^{a}	7	3.35^{a}	1.94^{a}
Threepoint Lake	5	1.16^{b}	1.01^{b}	4	0.67^{b}	0.74^{b}
Odei River	10	0.13^{b}	0.11^{c}	14	0.11^{b}	0.06^{c}

B. RED BACKED VOLES (Clethrionomys gapperi)

LOCATION	WATER EDGE			INLAND		
	N	Mean[1]	Median[2]	N	Mean[1]	Median[2]
Notigi Lake	3	0.20	0.24	6	0.16	0.15
Threepoint Lake	10	0.22	2.20	2	0.17	0.17

[1] Duncan's Multiple Range Test used to compare mean Hg levels.

[2] Mann-Whitney U Test used to compare median Hg levels.

NOTE: Within columns, values with different superscripts are significantly different from each other ($p < 0.05$).

Table 16. Production of mink and otter pelts from Registered Trapline Sections within the Agreement area[1]

AREA	YEAR	MINK	OTTER
Southern Indian Lake	1981/82	353	88
	1982/83	188	55
	1983/84	429	79
	1984/85	430	77
Nelson House	1981/82	315	76
	1982/83	74	48
	1983/84	8	4
	1984/85	374	70
Split Lake	1981/82	472	117
	1982/83	103	38
	1983/84	99	78
	1984/85	398	148
Pikwitonei	1981/82	273	62
	1982/83	94	37
	1983/84	69	59
	1984/85	306	101

---

[1] Source: Manitoba Department of Natural Resources

Table 17. Mercury concentrations (ug/g wet weight, standardized to 350 mm fork length)[1] in muscle of northern pike (Esox lucius) from different localities within the South Indian Lake (SIL), Nelson House (NEL) and Split Lake (SLK) Registered Trapline Sections. Weighted means for years for which mammal data are available are shown in bold print.

Location	Year							
	1978	1979	1980	1981	1982	1983	1984	1985
Channel	--	.63	.57	.62	.77	.60	--	.71
Camp 9	--	.57	.60	.66	.65	.65	.69	--
Region 4	--	.60	.63	.70	.61	--	.61	.68
Region 5	--	--	--	.69	.75	--	.61	--
Region 6	.61	.68	.75	.88	.92	.72	1.09	.65
Issett L.	.54	--	--	--	.76	.86	.86	.73
SIL overall (weighted mean)		.57			.75	.71	.78	
Rat L.	1.92	--	--	--	--	1.33	1.52	1.34
Notigi L.	--	--	2.58	1.48	2.24	1.23	--	1.39
Threepoint L	--	--	--	1.39	--	1.75	1.44	1.27
Wuskwatim L.	--	.82	.97	.85	.64	--	1.42	--
Mystery L.	--	.85	--	--	.86	.95	--	--
NEL overall (weighted mean)		.84				1.35	1.46	
Split Lake (SLK)	--	.32	--	.38	.50	.40	.31	--

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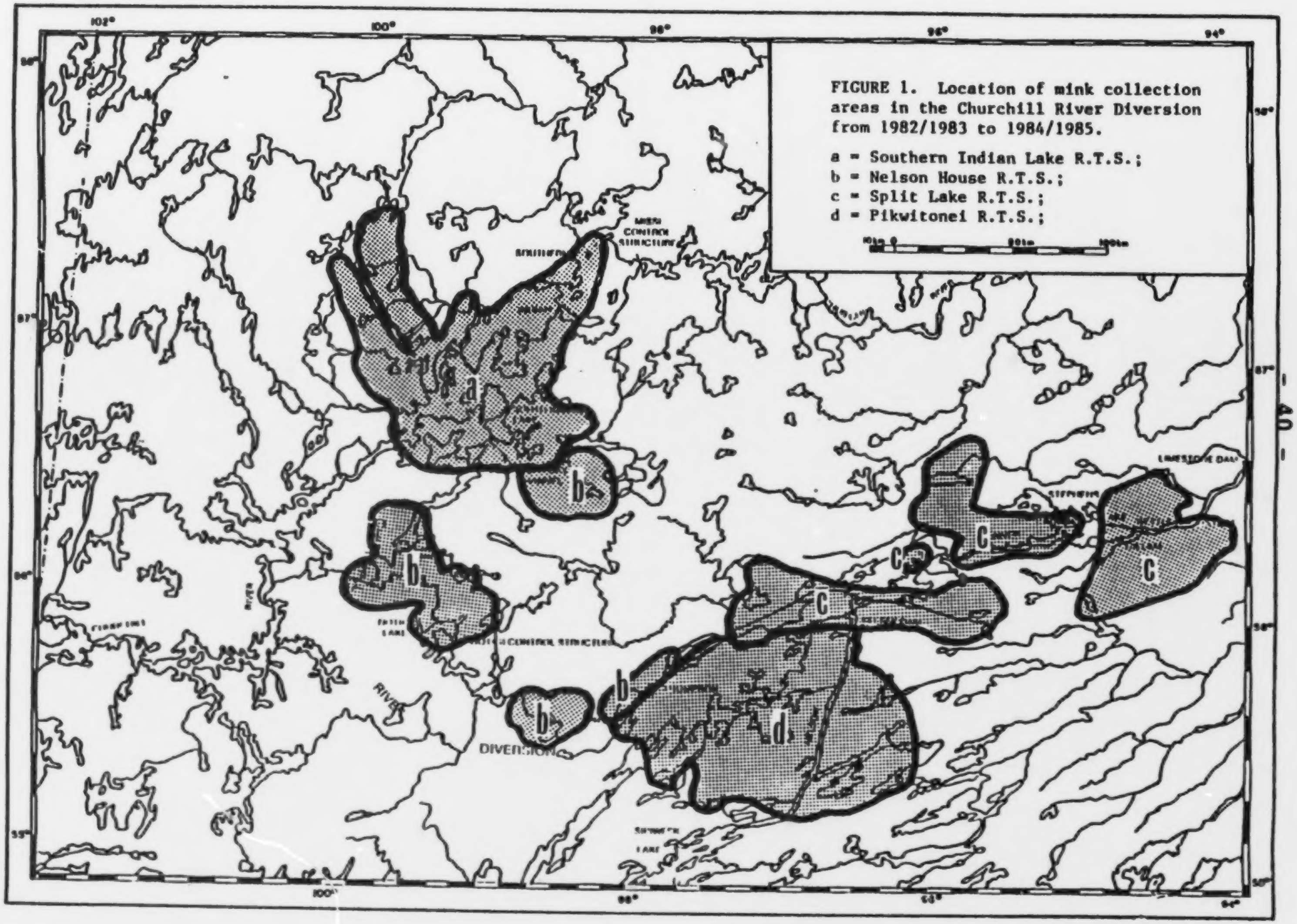
[1] Preliminary data supplied by R.A. Bodaly (Dept. Fisheries and Oceans), and J.O'Connor and D.Green (Manitoba Dept. Natural Resources).

FIGURES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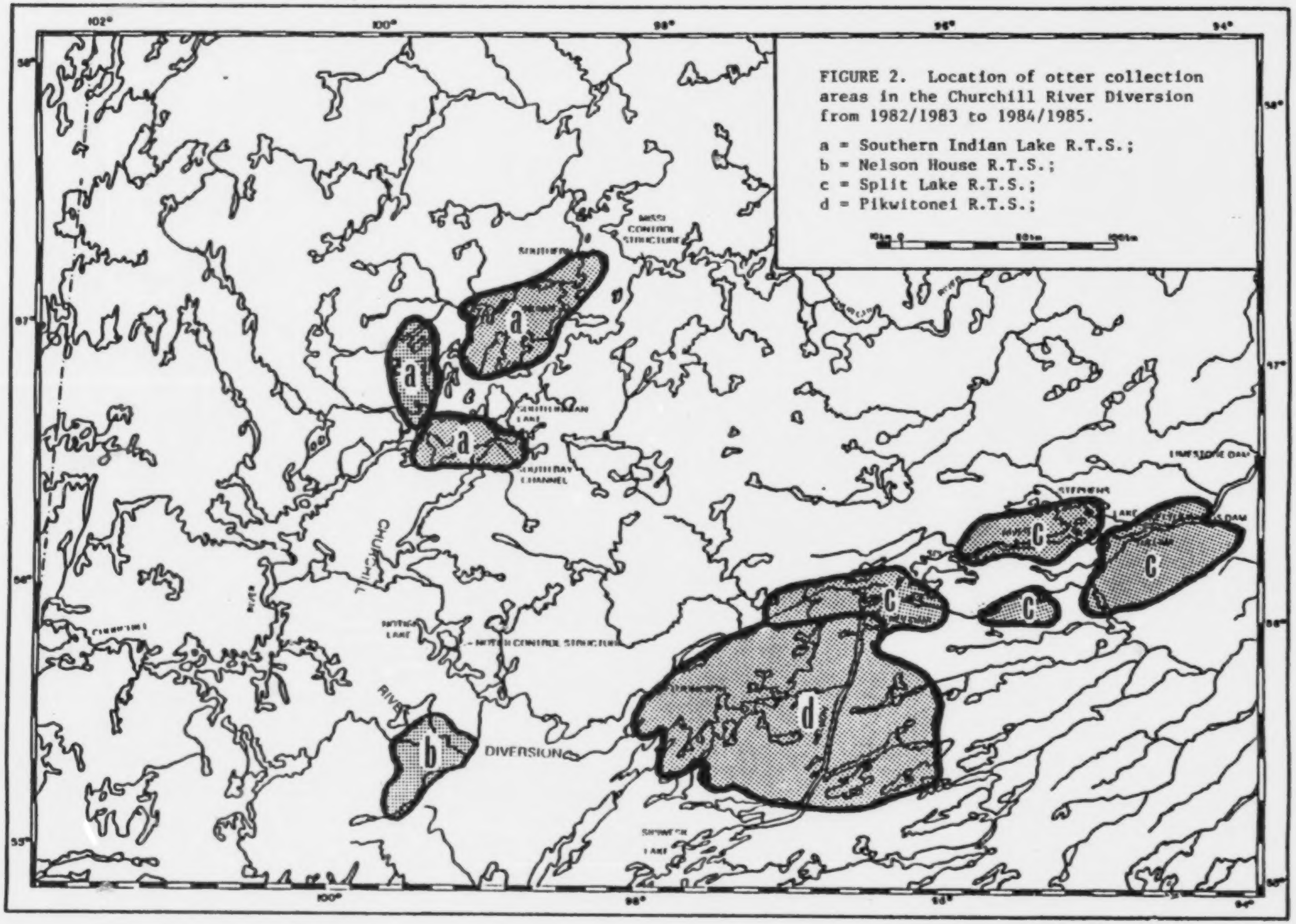
b
c
c
c
c
d
MISSI CONTROL STRUCTURE
RIVER
DIVERSION
LIMESTONE DAM

FIGURE 1. Location of mink collection areas in the Churchill River Diversion from 1982/1983 to 1984/1985.

a = Southern Indian Lake R.T.S.;
b = Nelson House R.T.S.;
c = Split Lake R.T.S.;
d = Pikwitonei R.T.S.;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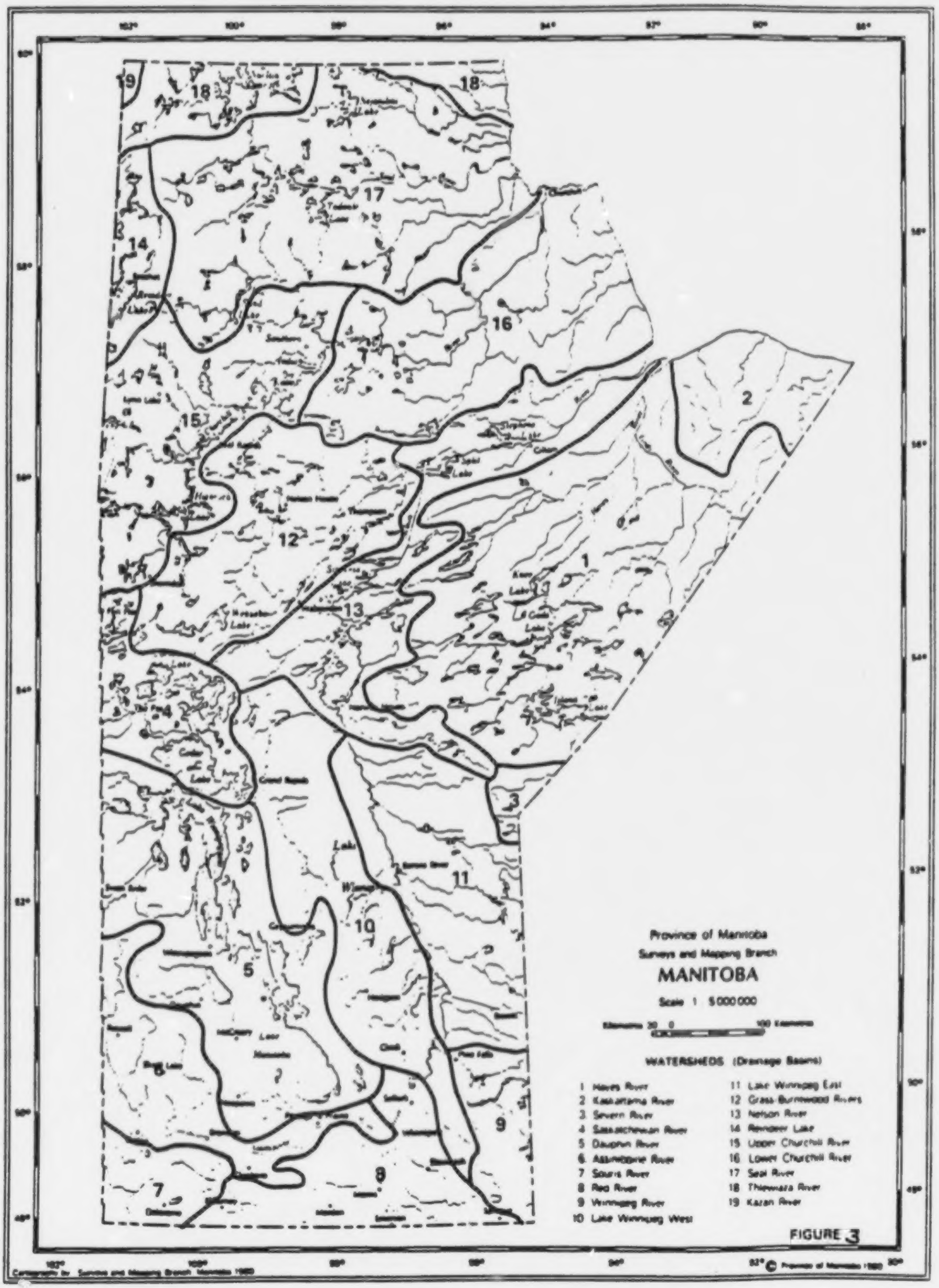
SOUTHERN INDIAN LAKE
SOUTH BAY CHANNEL
CHURCHILL
RIVER
DIVERSION
NOTIGI CONTROL STRUCTURE
LAKE
LIMESTONE DAM
STEPHENS
LAKE
DAM
a
a
a
b
c
c
c
c
d
10km 0
50km
100km

FIGURE 2. Location of otter collection areas in the Churchill River Diversion from 1982/1983 to 1984/1985.

a = Southern Indian Lake R.T.S.;
b = Nelson House R.T.S.;
c = Split Lake R.T.S.;
d = Pikwitonei R.T.S.;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Surveys and Mapping Branch
MANITOBA
Scale 1 : 5 000 000
Kilometres 20 0 100 Kilometres
WATERSHEDS (Drainage Basins)
1 Hayes River
2 Kaskattama River
3 Severn River
4 Saskatchewan River
5 Dauphin River
6 Assiniboine River
7 Souris River
8 Red River
9 Winnipeg River
10 Lake Winnipeg West
11 Lake Winnipeg East
12 Grass-Burntwood Rivers
13 Nelson River
14 Reindeer Lake
15 Upper Churchill River
16 Lower Churchill River
17 Seal River
18 Thlewiaza River
19 Kazan River
FIGURE 3
© Province of Manitoba 1980
Lake Winnipeg
Cedar Lake
Grand Rapids

Figure 3 Manitoba watersheds (drainage basins).

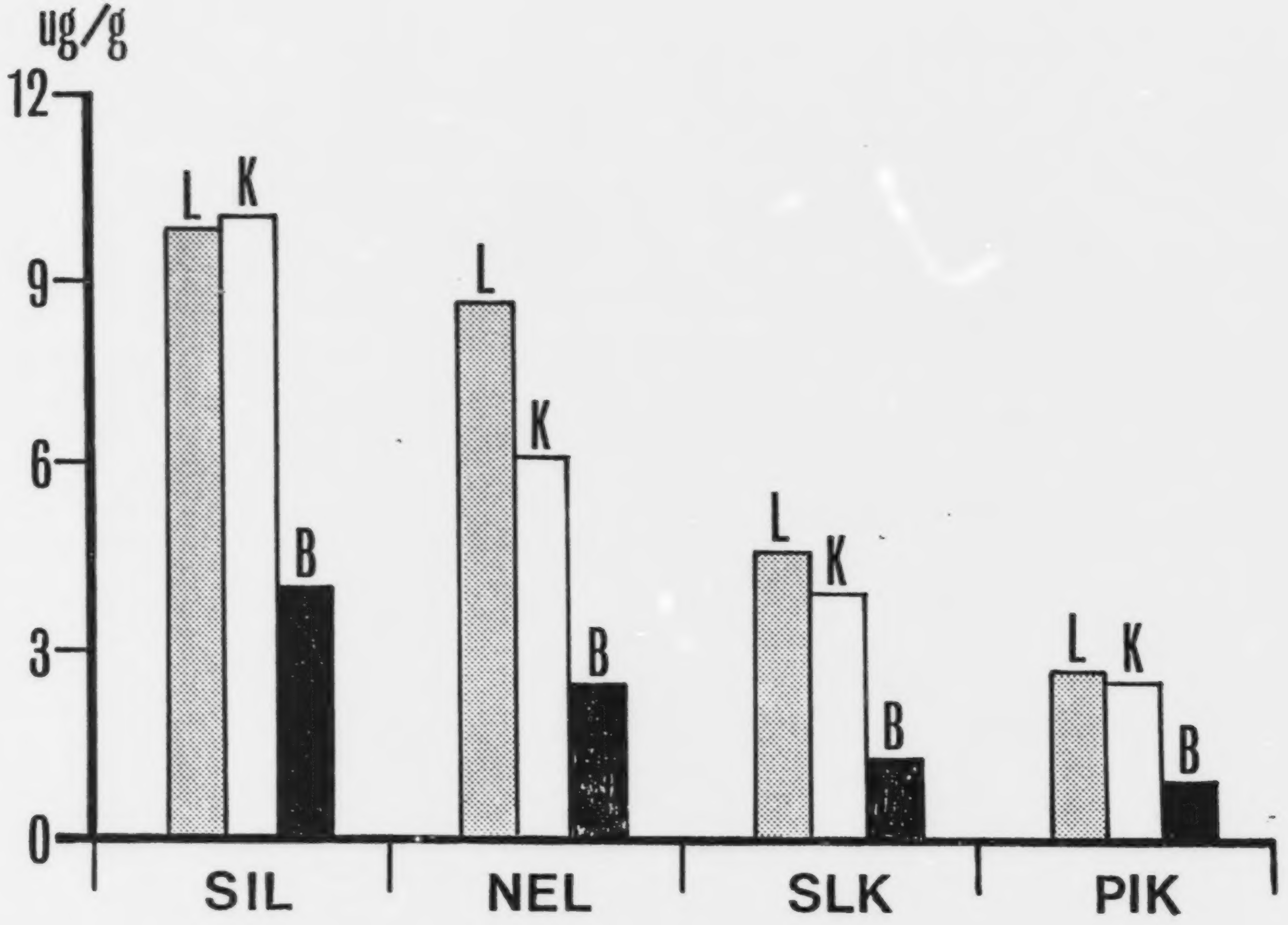

Figure 4 Mean concentrations of mercury in liver, kidney and brain tissues of mink from the Agreement area. L = liver; K = kidney; B = brain; SIL = Southern Indian Lake, NEL = Nelson House, SLK = Split Lake R.T.S.

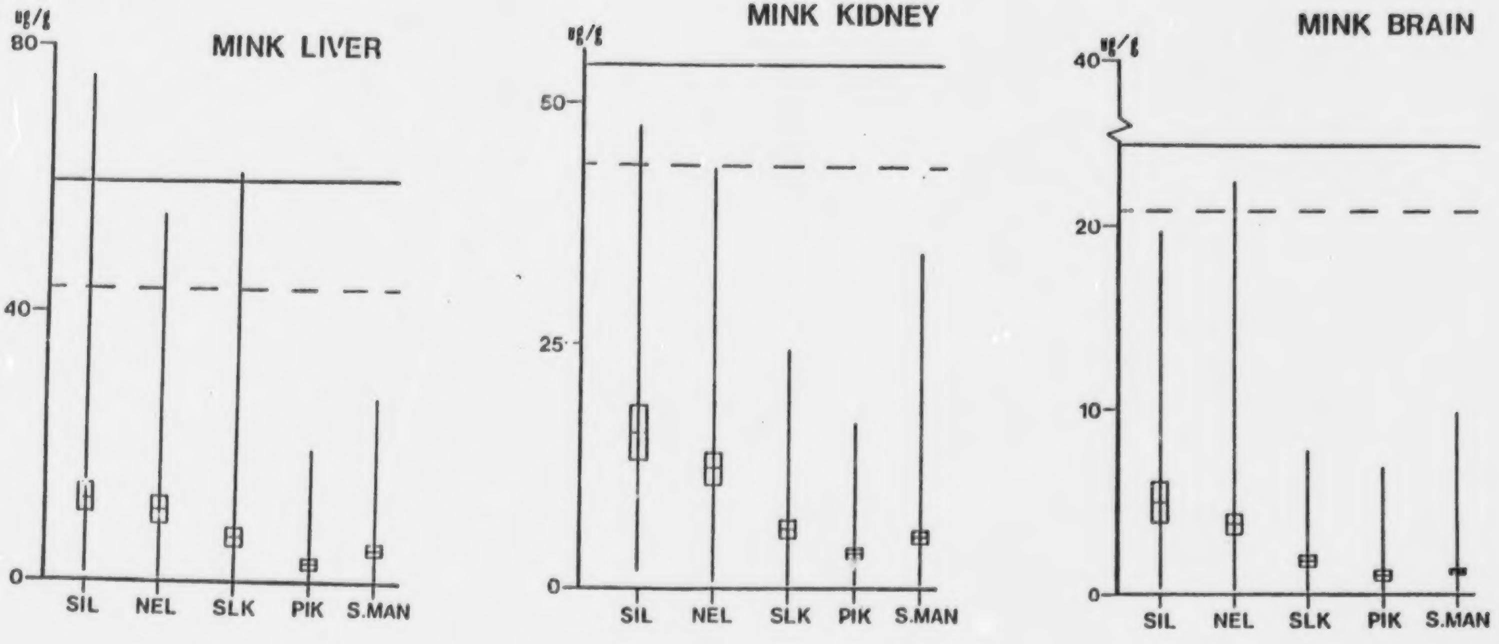

Figure 5. Mercury concentrations in tissues of Manitoba mink. Horizontal lines marke lethaly (solid line) and mildly (dashed line) toxic concentration for the respective tissue. SIL = Southern Indian Lake, NEL = Nelson House, SLK = Split Lake R.T.S.

APPENDIX A

Mercury Concentrations in Tissues of Mink

Descriptive statistics

Table A - 1 Descriptive statistics on mercury levels in brain tissue of mink from various areas of Manitoba (ug/g freeze-dried tissue).

AREA	YEAR	N	MEAN	MEDIAN	MODE	MINIMUM	MAXIMUM	S.D.	PERCENTILES			
									25th	50th	75th	95th
Southern Indian Lake	1984	18	5.36	4.38	0.29	0.29	19.7	4.95	1.99	4.38	7.77	19.70
	1985	9	4.30	2.49	0.79	0.79	16.2	4.75	1.63	2.49	5.46	16.20
Nelson House	1984	12	5.15	3.62	1.93	1.93	12.0	3.78	2.11	3.62	8.54	12.00
	1985	38	3.51	1.83	0.50	0.20	22.4	4.27	0.94	1.83	5.15	12.42
Split Lake	1983	13	2.49	2.29	0.34	0.34	6.45	1.80	1.04	2.29	3.59	6.45
	1934	14	3.17	2.33	0.98	0.96	7.75	2.36	0.98	2.33	5.69	7.75
	1985	43	1.23	0.93	0.47	0.11	4.48	0.99	0.48	0.93	1.67	3.40
Pikwitonei	1985	77	0.85	0.72	0.33	0.13	6.86	0.87	0.33	0.72	1.10	2.31
Southern Manitoba	1983	102	1.19	0.80	0.15	0.10	4.35	1.11	0.28	0.80	1.88	3.38
	1984	65	1.29	0.93	0.57	0.13	4.02	1.03	0.56	0.93	1.72	3.78
Winnipeg River	1982	11	2.68	2.04	0.90	0.90	9.81	2.57	1.09	2.04	3.30	9.81

Table A - 2 Descriptive statistics on mercury levels in liver tissue of mink from various areas of Manitoba (ug/g freeze-dried tissue).

AREA	YEAR	N	MEAN	MEDIAN	MODE	MINIMUM	MAXIMUM	S.D.	PERCENTILES 25th	PERCENTILES 50th	PERCENTILES 75th	PERCENTILES 95th
Southern Indian Lake	1984	18	16.73	10.90	2.09	2.09	74.7	18.97	4.92	10.90	17.25	74.70
	1985	9	13.05	7.18	3.97	3.97	43.0	12.58	5.24	7.18	18.55	43.00
Nelson House	1984	12	18.71	15.90	5.14	5.14	42.4	11.32	8.64	15.90	28.08	42.40
	1985	38	11.34	7.85	1.54	0.52	55.6	12.55	3.18	7.85	14.73	46.77
Split Lake	1983	13	9.21	8.68	1.40	1.40	18.5	5.73	3.15	8.68	14.20	18.50
	1984	12	10.37	8.10	2.32	2.32	26.3	8.19	3.19	8.10	17.15	26.30
	1985	43	5.82	3.63	0.37	0.37	62.1	9.82	1.44	3.63	6.53	21.40
Pikwitonei	1985	76	3.31	2.60	0.44	0.30	21.7	3.29	1.41	2.60	4.27	9.48
Southern Manitoba	1983	103	4.56	2.73	0.42	0.29	19.2	4.72	0.97	2.73	8.16	13.10
	1984	65	4.94	3.71	10.10	0.36	19.5	4.54	1.88	3.71	5.67	17.99
Winnipeg River	1982	11	7.34	2.67	0.46	0.46	29.4	8.49	2.36	2.67	12.00	29.40

Table A - 3 Descriptive statistics on mercury levels in kidney tissue of mink from various areas of Manitoba (ug/g freeze-dried tissue).

AREA	YEAR	N	MEAN	MEDIAN	MODE	MINIMUM	MAXIMUM	S.D.	PERCENTILES 25th	PERCENTILES 50th	PERCENTILES 75th	PERCENTILES 95th
Southern Indian Lake	1984	18	13.66	12.25	1.69	1.69	48.1	10.94	5.38	12.25	17.33	48.10
	1985	9	9.82	6.13	2.41	2.41	29.5	8.17	5.08	6.13	12.35	29.50
Nelson House	1984	12	10.89	9.49	4.46	4.46	23.6	6.67	5.23	9.49	15.38	23.60
	1985	38	8.94	5.06	0.59	0.59	39.0	9.03	2.72	5.06	12.35	28.36
Split Lake	1983	13	6.45	5.45	1.21	1.21	12.0	3.52	3.40	5.45	9.75	12.00
	1984	12	7.59	5.24	1.72	1.72	24.5	6.57	2.74	5.24	11.38	24.50
	1985	43	3.50	2.95	0.36	0.36	12.1	2.81	1.20	2.95	4.84	9.54
Pikwitonei	1985	76	2.65	2.19	1.19	0.16	13.2	2.04	1.22	2.19	3.40	6.55
Southern Manitoba	1983	103	3.66	2.44	0.70	0.39	13.2	3.20	1.18	2.44	5.53	10.10
	1984	65	3.96	3.18	2.09	0.56	13.7	2.87	1.83	3.18	5.49	9.89
Winnipeg River	1982	11	7.84	6.38	2.09	2.09	32.9	8.85	2.59	6.38	6.98	32.90

APPENDIX B

Mercury Concentrations in Tissues of Otter

Descriptive statistics

Table B - 1 Descriptive statistics on mercury levels in brain tissue of otter from various areas of Manitoba (ug/g freeze-dried tissue).

AREA	YEAR	N	MEAN	MEDIAN	MODE	MINIMUM	MAXIMUM	S.D.	PERCENTILES 25th	50th	75th	95th
Southern Indian Lake	1984	3	10.34	4.33	2.09	2.09	24.60	12.40	2.09	4.33	24.60	24.60
	1985	1	4.15	4.15	4.15	4.15	4.15	--	--	--	--	--
Nelson House	1984	2	4.08	4.08	2.56	2.56	5.60	2.15	2.56	4.08	5.60	5.60
Split Lake	1983	1	0.50	0.50	0.50	0.50	0.50	--	--	--	--	--
	1984	21	2.06	1.13	0.66	0.66	8.41	2.24	0.78	1.13	2.19	8.28
	1985	6	0.94	0.80	0.65	0.65	1.60	0.35	0.71	0.80	1.20	1.60
Pikwitonei	1985	19	0.96	0.80	0.35	0.35	2.64	0.62	0.47	0.80	1.18	2.64

Table B - 2 Descriptive statistics on mercury levels in liver tissue of otter from various areas of Manitoba (ug/g freeze-dried tissue).

AREA	YEAR	N	MEAN	MEDIAN	MODE	MINIMUM	MAXIMUM	S.D.	PERCENTILES 25th	50th	75th	95th
Southern Indian Lake	1984	3	13.20	17.10	5.20	5.20	17.30	6.93	5.20	17.10	17.30	17.30
	1985	1	11.60	11.60	11.60	11.60	11.60	--	--	--	--	--
Nelson House	1984	2	11.67	11.67	8.33	8.33	15.00	4.72	8.33	11.67	15.00	15.00
Split Lake	1983	1	3.03	3.03	3.03	3.03	3.03	--	--	--	--	--
	1984	7	18.31	12.10	3.28	3.28	43.00	15.81	4.74	12.10	37.50	43.00
	1985	6	6.06	4.32	2.01	2.01	17.80	5.85	2.71	4.32	8.20	17.80
Pikwitonei	1985	19	4.05	3.51	1.26	1.26	14.50	3.03	1.85	3.51	4.94	14.50
Mantario Lake (Winnipeg River drainage basin)	1983	2	11.49	11.49	6.58	6.58	16.40	6.94	6.58	11.49	16.40	16.40

Table B - 3 Descriptive statistics on mercury levels in kidney tissue of otter from various areas of Manitoba (ug/g freeze-dried tissue).

AREA	YEAR	N	MEAN	MEDIAN	MODE	MINIMUM	MAXIMUM	S.D.	PERCENTILES 25th	 50th	 75th	 95th
Southern Indian Lake	1984	3	17.54	15.30	6.13	6.13	31.20	12.69	6.13	15.30	31.20	31.20
	1985	1	21.60	21.60	21.60	21.60	21.60	--	--	--	--	--
Nelson House	1984	2	18.15	18.15	12.40	12.40	23.90	8.13	12.4	18.15	23.90	23.90
Split Lake	1983	1	3.04	3.04	3.04	3.04	3.04	--	--	--	--	--
	1984	7	12.98	7.37	2.84	2.84	35.70	12.52	3.27	7.37	24.6	35.70
	1985	6	5.71	3.20	2.18	2.18	14.40	5.00	2.19	3.20	10.44	14.40
Pikwitonei	1985	19	3.84	3.90	1.28	1.28	11.10	2.34	1.72	3.90	4.72	11.10
Mantario Lake (Winnipeg River drainage basin)	1983	2	5.21	5.21	4.99	4.99	5.42	0.30	4.99	5.21	5.42	5.42

Canada Manitoba

Canada-Manitoba Mercury Agreement

POSTSCRIPT PROJECT 3.1

The Technical Advisory Committee expressed the opinion that while mink can be used as mercury indicators, they cannot "serve as well as, or better than, fish." Their opinion was based on the uncertainty regarding the specific feeding areas and diet of the mink sampled in this study, and the apparently variable mink to fish mercury concentration ratio.

MERCURY DATA FROM INSPECTED COMMERCIAL SHIPMENTS OF FISH FROM THE CHURCHILL RIVER DIVERSION AREA IN MANITOBA

Inspection Branch
Central and Arctic Region
Fisheries and Oceans Canada
501 University Crescent
Winnipeg, Manitoba.

December 16, 1986.

SPECIAL NOTE

This report was prepared for the Canada-Manitoba Agreement on the Study and Monitoring of Mercury in the Churchill River Diversion. Any conclusions, recommendations, or opinions expressed herein are those of the authors and are not necessarily the same as those of the Steering Committee for this Agreement.

SUMMARY

The Inspection and Technology Branch inspects commercial shipments of fish to ensure unacceptable levels of mercury are not present in fish intended for export and inter-provincial sale.

Mercury levels in commercial shipments of fish from areas affected by the Churchill-Nelson River Diversion (Figure 1) appeared to increase significantly from 1976 to 1979 and then to stabilize at the higher levels. However, the severe adverse impact on the commercial fisheries which was forecast in 1978 due to the increasing mercury levels did not materialize due to a fortuitous action in the U.S.A.; the permitted mercury content in fish imported into that country was raised from 0.5 to 1.0 ppm total mercury. Most of the shipments from the commercial fisheries in which the mercury levels are known to have increased have mercury contents less than 1.0 ppm.

There are four areas directly on the Rat River where fish contain mercury levels consistently in excess of established tolerances. These lakes: Issett, Notigi, Rat, and Wapisu, were commercially fished prior to the diversion of the Churchill River, but they were not fished from 1971 to 1976 and the pre-impoundment mercury levels are unknown.

There do not appear to have been significant increases in the mercury contents of the commercial shipments of fish from the Nelson River, Split, and Sipiwesk Lakes. Mercury levels from these areas have been high since the mercury problem was discovered in 1969. The feasibility of existing commercial fishing operations from these areas would be even more threatened if the rate of rejection does increase.

There is a considerable range of mercury levels in fish in the lakes in the vicinity of the diversion. Most of the commercially fished lakes not directly affected by the flooding have acceptably low fish mercury levels, but Apussigamasi, Beatty, Beaucage, Hill, and Osik lakes contain mercury levels sufficiently high so that closure of the lakes to commercial fishing for N. pike and pickerel is recommended.

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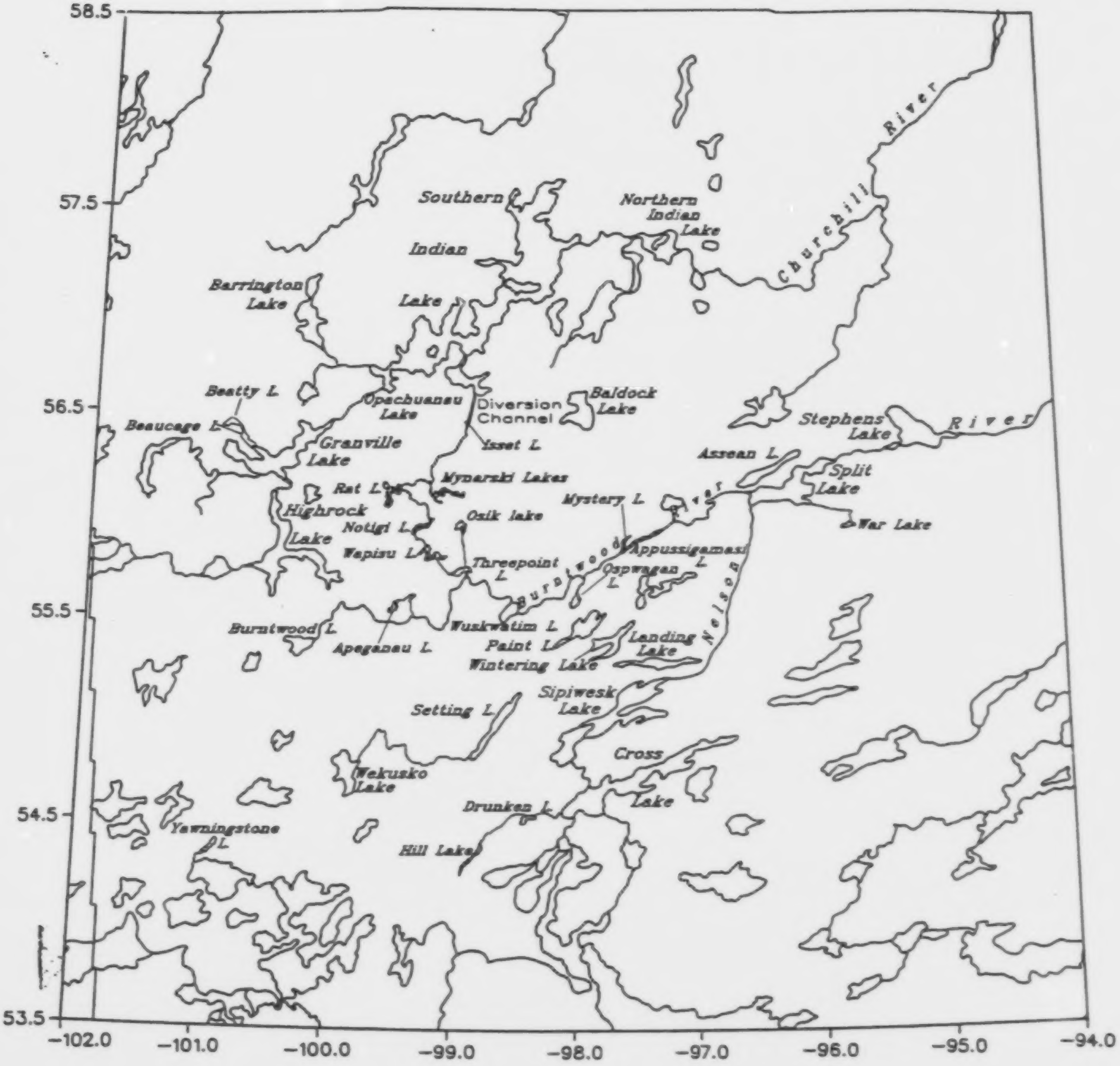
MAP: Churchill River Diversion Area iii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-94.0
Southern
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Lake
Northern
Indian
Lake
Churchill
River
Barrington
Lake
Beatty L.
Opachuanau
Lake
Diversion
Channel
Baldock
Lake
Beaucage L.
Granville
Lake
Isset L.
Stephens
Lake
River
Assean L.
Split
Lake
Rat L.
Mynarski Lakes
Mystery L.
River
War Lake
Highrock
Lake
Notigi L.
Osik lake
Wapisu L.
Threepoint
L.
Burntwood
Appussigamasi
L.
Ospwagan
L.
Nelson
Burntwood L.
Apeganau L.
Wuskwatim L.
Paint L.
Landing
Lake
Wintering Lake
Sipiwesk
Lake
Setting L.
Cross
Lake
Wekusko
Lake
Drunken L.
Yawningstone
L.
Hill Lake

FIGURE 1

CHURCHILL RIVER DIVERSION AREA

BACKGROUND

The Inspection Branch, Department of Fisheries & Oceans, is responsible for the inspection of all fish and fish products intended for export, including interprovincial sales. Since some freshwater species of fish from various areas accumulate mercury levels to concentrations in excess of allowable levels, Inspection Branch enforces a comprehensive mercury control program as part of its mandate.

The allowable level for total mercury in fish sold in Canada is 0.5 ppm. In the United States, where at least 80 percent of the commercial production from Northern Manitoba is marketed, the allowable tolerance for total mercury in fish was 0.5 ppm until 1979, when it was raised to 1.0 ppm. In 1984 the limit was again changed from 1.0 ppm total mercury to 1.0 ppm methyl mercury. In France, where a significant market for N. pike exists, the guideline for mercury was 0.7 ppm total mercury until 1982. Presently shipments exported to France must meet the requirements of the country of origin, which includes the Canadian 0.5 ppm tolerance.

All mercury analyses conducted on shipments of fish for commercial sale are for total mercury. Shipments which exceed the 0.5 ppm level are exported to the U.S.A. unless they are over 1.0 ppm. There is no current legal market for fish containing in excess of 1.0 ppm mercury and these fish are destroyed.

Commercial shipments from the the Rat River system were inspected for the first time in 1977 and 1978, and because of the high mercury levels there was a high rejection and destruction rate. Rat, Notigi, and Wapisu Lakes were found to be unusually high in comparison to normal mercury background levels in Manitoba. Beginning in 1976, samples of Barrington Lake N. pike, Drunken Lake N. pike and pickerel, Opachuanau Lake N. pike, and Southern Indian Lake N. pike and pickerel also began to frequently exceed the allowable mercury tolerances of 0.5 and 0.7 ppm and were subsequently destroyed. Prior to 1976 no shipments from these lakes were destroyed.

It appeared that the diversion of the Churchill River through the Rat River into the Nelson (which began in 1976) for the hydro-electric developments could be related to the increase in mercury levels along the entire system. In 1978 it seemed that if the trend towards increasing mercury levels continued, all commercial fishing along the diversion route would be threatened. These concerns were raised by the Inspection Branch and the discussions which resulted between the various Federal and Provincial departments and the Northern Flood Committee led to the formulation of the Canada-Manitoba Mercury Agreement.

The impact of the increased mercury levels on the commercial fisheries affected by the diversion was not as severe as initially forecasted. When the U.S.A. raised the allowable mercury limit in fish from 0.5 to 1.0 ppm the increase in tolerance paralleled the increase in mercury levels; all the production which would have been rejected at the 0.5 and 0.7 levels was still acceptable in the major market for those fish.

METHODS

1) Field Collection Program

Commercial shipments (lots of a particular species of fish of common form and size from a unique lake) are sampled at shipping points. The level of sampling employed is determined by the "lake classification system", a protocol whereby lakes have been grouped or classified according to the risk of certain species being rejected for high levels of mercury. This system is based on the historical mercury levels (data has been gathered since 1970) and is updated annually. Five classifications are defined as follows:

A - Monitor. No restrictions. Previous data indicates acceptable mercury levels and further sampling is at the discretion of inspection personnel.

B - Sample Each Shipment. No restrictions, but shipments should be sampled as often as possible because there is currently insufficient data to assign a valid lake classification, or, there are indications the mercury levels may be changing.

C - Closed. These lakes have either been closed by legislative action or have been recommended for closure by the Inspection Branch. The estimated rejection rates are considered too high for fishing from these lakes to be feasible. If production is received from these areas, detention procedures should be followed as described below.

D - Detain Each Shipment. These shipments have a high risk of rejection even at the 1.0 ppm guideline and must be detained and sampled to ensure acceptable mercury levels prior to marketing for any country. Shipments in excess of 1.0 ppm are to be destroyed.

E - Export. Species with this classification cannot be marketed in Canada or Europe unless they are sampled and found to be within the 0.5 tolerance. These shipments can be marketed to the U.S.A. without restriction and sampling.

Species from problem areas, that is, those lakes classified as "export", are sampled at least three times a year to ensure that any changes in mercury levels are identified on a timely basis.

Species which originate from areas for which there is no data must be sampled for three consecutive shipments to establish a tentative classification.

The minimum number of specimens considered to be representative of a commercial shipment is five fish, and the sample weight must be a minimum of 7 kg in total. The fish are selected from the lot at random.

Generally, only N. pike (Esox lucius), yellow pickerel (Stizostedion vitreum), and lake trout (Salvelinus namaycush) are classified, as they are the only commercially important species which may have a significant risk of rejection.

2) **Laboratory Methods**

To prepare samples for mercury analyses a skinned fillet is taken from each fish in the sample and these are ground together to obtain a homogeneous composite sample. The composite samples are analyzed in triplicate for total mercury. Fish tissue is solubilized using concentrated nitric and sulfuric acids and heated at 180 degrees C overnight. The following day the mercury, which is then in ionic form, is reduced with stannous sulfate, partitioned into air, and analyzed by cold vapor atomic absorption spectroscopy. Two "in-house" quality control samples are analyzed in triplicate with each batch of samples. Mercury levels of these samples are just above and just below the Canadian guideline. In addition, the laboratory coordinates and participates in an international Mercury Quality Assurance Program. Three times per year, sets of four samples of canned fish covering a range of mercury values are distributed to approximately twenty-five participating laboratories.

DISCUSSION OF RESULTS

Summaries of mercury inspection results provided by lake, species and year are in Table 1. The lakes in Table 1 include all the areas commercially fished which were directly affected by the diversion, commercially fished lakes not directly affected but in the general diversion area, and lakes which were identified as domestic food sources by communities in the Diversion area. The information provided in Table 1 includes the lake name and coordinates, the species, the year, the number of composite samples, the mean ppm for the samples, the range in ppm of the samples for the given year, and the percentage of the number of samples which exceeded the 0.5 ppm and 1.0 ppm tolerances. Species are asterisked where analyses of variance tests indicate the differences in means between years are significant.

The data analysed in this report is from inspection samples only. The results from the pooled sampling method are utilized to estimate the average mercury contents of shipments from given lakes and species. The method is not designed to provide estimates of the baseline mercury levels in the fish populations of a lake.

The data for N. pike should be treated with some caution. It is common practice for shipments of N. pike from these lakes to be graded and packed by size at the lake level. The mercury levels in the large fish are highest and have the highest risk of rejection, so samples taken from the larger size grades are sampled more frequently, even though the larger fish comprise a relatively small portion of most shipments. The summaries in Table 1 for some of these lakes will reflect the bias since the disproportionate number of samples from large fish is not representative of the actual commercial production. The data in Tables 2 and 3 provides a better indication of the impact of high mercury levels on a particular fishery.

Tables 2 and 3 provide the amount of fish rejected (marketed weight) for each lake and species in the general diversion area from 1977 to 1984, The landed production (in terms of round equivalent weight) from each lake and species where there has been a rejection is also given. For Lake Mynarski West, Rat Lake and Wapisu Lake, the production figures from 1962 to 1985 are given.

Mercury levels in N. pike and pickerel from Southern Indian Lake increased in 1976. This increase was also observed in other lakes such as Barrington, Drunken, and Opachuanau Lakes. The increase in mercury levels appeared quite rapid, but by 1979 the mercury levels appeared to have stabilized at the higher levels in the fish. This rise in mercury levels would have had severe adverse effects on commercial fisheries if not for the action by the U.S.A. in 1979 which raised their allowable mercury content in fish from 0.5 to 1.0 ppm. At the current levels there remains a problem with the acceptability of N. pike from Southern Indian Lake. In the 1984/85 summer and winter fishing seasons, 18,200 kg of N. pike were inspected and 567 kg were destroyed due to excessive mercury contents (Table 2).

The mercury levels in fish from the Nelson River System did not appear to change as a result of the diversion. The mercury in N. pike and pickerel from the Nelson River, Split and Sipiwesk lakes were seriously high at the outset of our sampling programs and have not appeared to have exhibited any significant increases or decreases. Prior to 1979 these lakes were fished primarily for whitefish which contain much lower mercury levels, and for N. pike, which if acceptable, was directed at the markets in France which at that time had a mercury tolerance of 0.7 ppm, but there were substantial rates of rejection. Any increases in the mercury levels in the fish and rates of rejection from these lakes would probably mean the end of viable commercial fisheries.

Four lakes on the Rat River system: Issett, Notigi, Rat and Wapisu Lakes, are presently not commercially fished due to high mercury levels in the fish.

There are a considerable number of lakes in the diversion area but not directly on the diversion route which appear to have unusually high background levels for Manitoba. Appussigamasi, Beatty, Beaucage, Hill, and Osik lakes are recommended for closure due to high mercury levels in the fish.

Drunken Lake appeared to exhibit the same rise in mercury levels as some of the lakes on the Diversion route even though it is not on the system. More comprehensive testing of fish from this lake is required to confirm or explain the apparent variations in the mercury levels.

ACKNOWLEDGEMENTS:

The commercial fish production figures used in this report were taken from Freshwater Fish Marketing Corporation reports, or obtained from the Manitoba Department of Natural Resources in The Pas, Manitoba. Assistance was provided by Robert Whiteford in obtaining information.

We thank L. McKerness for her assistance and patience in waiting for this report.

TABLE 1 - SUMMARIES BY LAKE, SPECIES, YEAR OF MERCURY INSPECTION DATA

LAKE NAME & CODE #	SPECIES	YEAR	N	MEAN PPM	LOW PPM	HIGH PPM	% REJECTION	
							0.5	1.0
Apeganau 5535 9935	Pike, N.*	1976	2	0.84	0.84	0.85	100	0
		1977	1	1.03	1.03	1.03	100	100
		ALL	3	0.91	0.84	1.03	100	33
	Pickerel*	1976	5	0.55	0.43	0.63	60	0
		1977	3	0.35	0.31	0.39	0	0
		ALL	8	0.47	0.31	0.63	38	0
Apussigamasi 5551 9736	Pike, N.	1982	2	1.25	0.66	1.84	100	50
		1983	1	1.52	1.52	1.52	100	100
		1984	6	1.70	1.23	2.32	100	100
		ALL	9	1.58	0.66	2.32	100	89
	Pickerel	1982	1	2.37	2.37	2.37	100	100
		1983	1	1.75	1.75	1.75	100	100
		ALL	2	2.06	1.75	2.37	100	100
Assean 5613 9630	Pike, N.	1982	1	0.57	0.57	0.57	100	0
		1983	1	0.42	0.42	0.42	0	0
		ALL	2	0.49	0.42	0.57	50	0
	Pickerel	1978	3	0.32	0.28	0.40	0	0
		1982	1	0.32	0.32	0.32	0	0
		1985	1	0.29	0.29	0.29	0	0
		ALL	5	0.32	0.28	0.40	0	0
Baldock 5633 9757	Pike, N.	1972	1	0.17	0.17	0.17	0	0
		1979	1	0.34	0.34	0.34	0	0
		1980	4	0.46	0.38	0.62	25	0
		1982	2	0.38	0.36	0.40	0	0
		ALL	8	0.39	0.17	0.62	13	0
	Pickerel	1970	1	0.27	0.27	0.27	0	0
		1972	1	0.18	0.18	0.18	0	0
		1979	1	0.34	0.34	0.34	0	0
		1980	4	0.39	0.30	0.43	0	0
		1982	2	0.33	0.24	0.42	0	0
		ALL	9	0.33	0.18	0.43	0	0
	Trout, L.	1970	1	0.24	0.24	0.24	0	0
		1972	1	0.32	0.32	0.32	0	0
		1980	2	0.16	0.14	0.19	0	0
		ALL	4	0.22	0.14	0.32	0	0

* analyses of variance indicates differences in the mean mercury levels between years are significant (P >.95).

LAKE NAME	SPECIES	YEAR	N	MEAN PPM	LOW PPM	HIGH PPM	% REJECTION	
							0.5	1.0
Barrington 5655 10015	Pike, N.*	1970	1	0.36	0.36	0.36	0	0
		1971	2	0.41	0.37	0.45	0	0
		1974	1	0.33	0.33	0.33	0	0
		1975	1	0.41	0.41	0.41	0	0
		1976	9	0.51	0.38	0.67	11	0
		1977	2	0.63	0.59	0.67	100	0
		1978	1	0.75	0.75	0.75	100	0
		1983	1	0.45	0.45	0.45	0	0
		1985	1	0.48	0.48	0.48	0	0
		ALL	19	0.50	0.33	0.75	21	0
	Pickerel	1970	1	0.25	0.25	0.25	0	0
		1971	1	0.13	0.13	0.13	0	0
		1974	1	0.26	0.26	0.26	0	0
		1975	1	0.40	0.40	0.40	0	0
		1976	3	0.43	0.39	0.47	0	0
		1977	1	0.42	0.42	0.42	0	0
		1978	1	0.46	0.46	0.46	0	0
		1982	1	0.37	0.37	0.37	0	0
		1983	1	0.28	0.28	0.28	0	0
		1985	1	0.33	0.33	0.33	0	0
		ALL	12	0.35	0.13	0.47	0	0
Beatty 5631 10032	Pike, N.	1984	5	1.02	0.33	1.41	80	60
	Pickerel	1984	3	1.31	1.22	1.38	100	100
Beaucage 5627 10034	Pike, N.	1979	2	1.30	1.08	1.52	100	100
		1982	4	1.01	0.74	1.42	100	25
		1984	3	1.13	1.00	1.24	100	67
		ALL	9	1.11	0.74	1.52	100	56
	Pickerel	1979	2	1.23	1.13	1.33	100	100
		1982	4	1.13	0.93	1.40	100	75
		1984	3	1.03	0.66	1.32	100	67
		ALL	9	1.12	0.66	1.40	100	78
Burntwood L 5522 10026	Pike, N.	1971	4	0.36	0.21	0.48	0	0
		1972	5	0.41	0.29	0.61	20	0
		1976	2	0.49	0.47	0.51	0	0
		1977	1	0.41	0.41	0.41	0	0
		1979	1	0.29	0.29	0.29	0	0
		1981	2	0.55	0.54	0.56	50	0
		ALL	15	0.42	0.21	0.61	13	0

* analyses of variance indicates differences in the mean mercury levels between years are significant (P >.95).

LAKE NAME	SPECIES	YEAR	N	MEAN PPM	LOW PPM	HIGH PPM	% REJECTION	
							0.5	1.0
Burntwood L 5522 10025	Pickerel	1971	10	0.26	0.13	0.37	0	0
		1972	4	0.31	0.25	0.37	0	0
		1973	1	0.20	0.20	0.20	0	0
		1974	1	0.22	0.22	0.22	0	0
		1976	2	0.30	0.29	0.31	0	0
		1977	1	0.23	0.23	0.23	0	0
		1979	1	0.37	0.37	0.37	0	0
		1981	2	0.28	0.26	0.30	0	0
		ALL	22	0.28	0.13	0.37	0	0
Burntwood River 5608 9634	Pike, N.	1980	1	1.16	1.16	1.16	100	100
	Pickerel	1980	1	1.41	1.41	1.41	100	100
	Sauger	1980	1	1.66	1.66	1.66	100	100
	Maria (burbot)	1980	1	0.83	0.83	0.83	100	0
Drunken 5431 9827	Pike, N.*	1970	2	0.63	0.60	0.66	100	0
		1971	4	0.55	0.38	0.50	50	0
		1972	6	0.48	0.34	0.73	33	0
		1973	1	0.31	0.31	0.31	0	0
		1974	2	0.31	0.30	0.32	0	0
		1975	12	0.37	0.17	0.58	25	0
		1976	8	0.60	0.47	0.86	50	0
		1977	1	0.77	0.77	0.77	100	0
		1979	2	1.20	1.13	1.28	100	100
		1981	5	1.11	0.59	1.67	100	40
		1982	1	0.37	0.37	0.37	0	0
		1983	3	1.04	0.81	1.35	100	33
		1984	2	0.64	0.52	0.77	50	0
		1985	2	0.58	0.45	0.72	50	0
		ALL	51	0.61	0.17	1.67	51	10
	Pickerel*	1970	2	0.60	0.56	0.65	100	0
		1971	5	0.36	0.38	0.42	0	0
		1972	7	0.35	0.22	0.56	14	0
		1973	1	0.49	0.49	0.49	0	0
		1974	2	0.56	0.45	0.67	50	0
		1975	9	0.80	0.64	0.94	100	0
		1976	9	0.90	0.78	1.05	100	11
		1977	1	1.15	1.15	1.15	100	100
		1979	1	0.57	0.57	0.57	100	0
		1981	4	0.61	0.44	0.83	50	0
		1982	1	0.61	0.61	0.61	100	0
		1983	1	0.64	0.64	0.64	100	0
		1985	1	0.61	0.61	0.61	100	0
		ALL	44	0.64	0.22	1.15	66	5

* analyses of variance indicates differences in the mean mercury levels between years are significant (P >.95).

LAKE NAME	SPECIES	YEAR	N	MEAN PPM	LOW PPM	HIGH PPM	% REJECTION	
							0.5	1.0
Granville 5618 10030	Pike, N.	1971	4	0.43	0.14	0.64	50	0
		1973	1	0.36	0.36	0.36	0	0
		1974	1	0.22	0.22	0.22	0	0
		1976	3	0.40	0.31	0.52	0	0
		1977	4	0.43	0.14	0.64	50	0
		1973	1	0.36	0.36	0.36	0	0
		1974	1	0.22	0.22	0.22	0	0
		1979	1	0.65	0.65	0.65	100	0
		1980	1	0.61	0.61	0.61	100	0
		1982	1	0.62	0.62	0.62	100	0
		1985	1	0.41	0.41	0.41	0	0
		ALL	15	0.44	0.14	0.65	33	0
	Pickerel*	1970	1	0.39	0.39	0.39	0	0
		1971	7	0.22	0.16	0.32	0	0
		1972	1	0.10	0.10	0.10	0	0
		1973	2	0.22	0.22	0.22	0	0
		1974	1	0.30	0.30	0.30	0	0
		1975	1	0.28	0.28	0.28	0	0
		1976	2	0.28	0.28	0.28	0	0
		1977	1	0.29	0.29	0.29	0	0
		1978	1	0.23	0.23	0.23	0	0
		1980	1	0.30	0.30	0.30	0	0
		1982	1	0.41	0.41	0.41	0	0
		1985	1	0.22	0.22	0.22	0	0
		ALL	21	0.26	0.10	0.41	0	0
Hill 5420 9853	Pike, N.	1971	1	0.80	0.80	0.80	100	0
		1977	4	1.19	0.54	1.72	75	75
		ALL	5	1.11	0.54	1.72	80	60
	Pickerel	1977	4	1.16	0.93	1.46	100	50
Issett 5637 9909	Pike, N.	1984	6	1.08	0.80	1.38	100	50
		1985	1	1.58	1.58	1.58	100	100
		ALL	7	1.15	0.80	1.58	100	57
	Pickerel	1984	3	1.33	1.23	1.46	100	100
		1985	1	1.13	1.13	1.13	100	100
		ALL	4	1.28	1.13	1.46	100	100
Mynarski East 5610 9912	Whitefish	1980	2	0.22	0.21	0.23	0	0
	Pike, N.*	1977	4	1.08	0.71	1.38	100	50
		1980	1	1.31	1.31	1.31	100	100
		1981	2	0.37	0.36	0.38	0	0
		1982	3	0.87	0.50	1.48	67	33
		1983	2	0.37	0.33	0.41	0	0
		1984	5	0.39	0.32	0.50	0	0
		ALL	17	0.69	0.32	1.48	41	24

* analyses of variance indicates differences in the mean mercury levels between years are significant (P >.95).

LAKE NAME	SPECIES	YEAR	N	MEAN PPM	LOW PPM	HIGH PPM	% REJECTION	
							0.5	1.
Mynarski East 5610 9912	Pickerel*	1977	4	0.84	0.69	1.19	100	25
		1980	1	1.71	1.71	1.71	100	100
		1981	8	0.57	0.34	0.96	38	0
		1982	3	0.44	0.33	0.53	0	0
		1983	2	0.53	0.36	0.71	50	0
		1985	1	0.42	0.42	0.42	0	0
		ALL	19	0.65	0.33	1.71	47	11
	Tullibee	1980	2	0.45	0.37	0.54	0	0
Mystery 5551 9744	Pike, N.	1982	6	1.09	0.58	1.44	100	50
	Pickerel	1982	1	0.99	0.99	0.99	100	0
	Sturgeon	1982	1	0.20	0.20	0.20	0	0
Nelson River 5704 9230	Pike, N.	1975	3	0.79	0.72	0.88	100	0
		1976	10	1.01	0.74	1.47	100	40
		1978	4	1.09	0.98	1.22	100	75
		1979	5	1.03	0.58	1.45	100	60
		1980	3	0.82	0.66	0.93	100	0
		1981	2	0.70	0.62	0.78	100	0
		1982	3	1.08	0.99	1.13	100	67
		1983	6	0.90	0.52	1.25	83	17
		1984	14	0.83	0.43	1.65	64	21
		1985	6	0.88	0.54	1.41	83	33
		ALL	56	0.96	0.43	1.65	88	32
	Pickerel	1975	4	0.75	0.56	0.91	100	0
		1976	11	0.88	0.45	1.26	82	27
		1977	1	0.67	0.67	0.67	100	0
		1978	2	0.46	0.27	0.66	50	0
		1979	5	0.67	0.53	0.86	80	0
		1980	2	0.48	0.44	0.53	0	0
		1981	2	0.67	0.60	0.74	100	0
		1982	1	0.57	0.57	0.57	100	0
		1983	1	0.51	0.51	0.51	0	0
		1985	1	0.63	0.63	0.63	100	0
		ALL	30	0.72	0.27	1.26	77	10
	Sturgeon	1970	5	0.10	0.06	0.13	0	0
		1975	1	0.06	0.06	0.06	0	0
		1976	1	0.16	0.16	0.16	0	0
		1977	1	0.06	0.06	0.06	0	0
		1978	1	0.10	0.10	0.10	0	0
		1982	1	0.12	0.12	0.12	0	0
		1983	1	0.12	0.12	0.12	0	0
		ALL	11	0.10	0.06	0.16	0	0

* analyses of variance indicates differences in the mean mercury levels between years are significant (P >.95).

LAKE NAME	SPECIES	YEAR	N	MEAN PPM	LOW PPM	HIGH PPM	% REJECTION	
							0.5	1.0
Notigi 5557 9918	Whitefish	1980	1	0.12	0.12	0.12	0	0
	Pike, N.	1977	1	1.59	1.59	1.59	100	100
		1980	1	1.99	1.99	1.99	100	100
		1982	2	2.06	1.91	2.21	100	100
		ALL	4	1.92	1.59	2.21	100	100
	Pickerel	1978	3	1.32	0.26	2.32	67	67
		1980	1	2.59	2.59	2.59	100	100
		1982	1	1.11	1.11	1.11	100	100
		ALL	5	1.53	0.26	2.59	80	80
Opachuanau 5644 9937	Whitefish	1970	1	0.06	0.06	0.06	0	0
		1976	1	0.05	0.05	0.05	0	0
		1981	1	0.13	0.13	0.13	0	0
		ALL	3	0.08	0.05	0.13	0	0
	Pike, N	1971	6	0.31	0.22	0.45	0	0
		1973	1	0.31	0.31	0.31	0	0
		1976	1	0.60	0.60	0.60	100	0
		1977	1	0.53	0.53	0.53	0	0
		1978	3	0.65	0.38	0.97	67	0
		1979	1	0.20	0.20	0.20	0	0
		1985	1	1.01	1.01	1.01	100	100
		ALL	14	0.46	0.20	1.01	29	7
	Pickerel*	1970	1	0.20	0.20	0.20	0	0
		1971	6	0.18	0.12	0.22	0	0
		1973	1	0.21	0.21	0.21	0	0
		1975	1	0.24	0.24	0.24	0	0
		1976	1	0.38	0.38	0.38	0	0
		1977	1	0.30	0.30	0.30	0	0
		1978	1	0.35	0.35	0.35	0	0
		1979	1	0.72	0.72	0.72	100	0
		1980	1	0.51	0.51	0.51	0	0
		1985	1	0.47	0.47	0.47	0	0
		ALL	15	0.30	0.12	0.72	7	0
Osik 5598 9858	Pike, N.	1982	1	1.32	1.32	1.32	100	100
		1983	1	1.83	1.83	1.83	100	100
		1984	1	1.43	1.43	1.43	100	100
		ALL	3	1.53	1.32	1.83	100	100
	Pickerel	1982	1	1.05	1.05	1.05	100	100
		1983	3	0.57	0.09	0.86	67	0
		1984	1	0.99	0.99	0.99	100	0
		1985	3	0.69	0.52	1.00	33	0
		ALL	8	0.73	0.09	1.05	63	13

* analyses of variance indicates difference in the mean mercury levels between years are significant (P >.95).

LAKE NAME	SPECIES	YEAR	N	MEAN PPM	LOW PPM	HIGH PPM	% REJECTION	
							0.5	1.0
Rat 5610 9940	Whitefish	1978	1	0.37	0.37	0.37	0	0
		1980	1	0.34	0.34	0.34	0	0
		ALL	2	0.35	0.34	0.37	0	0
	Pike, N	1978	5	2.14	2.04	2.25	100	100
		1980	1	2.32	2.32	2.32	100	100
		ALL	6	2.17	2.04	2.32	100	100
	Pickerel*	1978	5	2.54	2.39	2.67	100	100
		1980	1	1.15	1.15	1.15	100	100
		ALL	6	2.31	1.15	2.67	100	100
	Tullibee	1980	1	0.47	0.47	0.47	0	0
Sipiwesk 5505 9735	Whitefish	1970	3	0.17	0.13	0.21	0	0
	Pike, N	1970	15	0.89	0.42	1.71	93	27
		1974	9	0.77	0.60	0.94	100	0
		1975	12	1.03	0.39	1.50	92	52
		1978	4	0.67	0.60	0.71	100	0
		1979	6	1.06	0.66	1.58	100	50
		1980	13	0.88	0.62	1.73	100	23
		1981	55	1.01	0.16	1.53	91	31
		1982	43	0.95	0.25	2.31	91	44
		1983	15	0.95	0.43	1.90	87	33
		1984	13	0.76	0.45	1.30	77	8
		1985	7	1.01	0.07	1.52	86	57
		ALL	192	0.92	0.07	2.31	91	32
	Pickerel*	1970	12	0.88	0.69	1.11	100	17
		1974	9	0.67	0.58	0.77	100	0
		1975	18	0.74	0.49	0.92	94	0
		1978	1	0.63	0.63	0.63	100	0
		1979	9	0.78	0.24	1.10	78	11
		1980	8	0.88	0.47	1.18	88	25
		1985	2	0.47	0.39	0.55	50	0
		ALL	108	0.74	0.12	1.32	82	12
	Sturgeon	1978	1	0.06	0.06	0.06	0	0
	Goldeye	1970	9	0.27	0.14	0.60	11	0
		1974	1	0.23	0.23	0.23	0	0
		ALL	10	0.26	0.14	0.60	10	0

* analyses of variance indicates differences in the mean mercury levels between years are significant (P >.95).

LAKE NAME	SPECIES	YEAR	N	MEAN PPM	LOW PPM	HIGH PPM	% REJECTION 0.5	1.0
Southern Indian 5716 9830	Whitefish*	1970	1	0.02	0.02	0.02	0	0
		1972	1	0.11	0.11	0.11	0	0
		1973	2	0.02	0.02	0.02	0	0
		1980	7	0.09	0.07	0.10	0	0
		ALL	11	0.07	0.02	0.11	0	0
	Pike, N.*	1971	4	0.26	0.24	0.29	0	0
		1972	5	0.32	0.24	0.40	0	0
		1973	3	0.30	0.26	0.33	0	0
		1976	10	0.47	0.26	1.02	10	10
		1977	2	0.43	0.42	0.45	0	0
		1978	7	0.50	0.25	0.83	29	0
		1979	9	0.88	0.53	1.51	89	22
		1980	14	0.67	0.45	1.12	71	7
		1981	39	0.91	0.53	2.04	95	21
		1982	17	0.90	0.25	1.99	94	35
		1983	26	0.81	0.32	1.27	96	15
		1984	15	0.97	0.63	1.63	100	33
		1985	1	0.81	0.81	0.81	100	0
		ALL	152	0.77	0.24	2.04	76	18
	Pickerel*	1971	6	0.19	0.16	0.22	0	0
		1972	3	0.18	0.18	0.23	0	0
		1973	3	0.28	0.20	0.35	0	0
		1975	2	0.30	0.22	0.38	0	0
		1976	4	0.24	0.20	0.32	0	0
		1977	2	0.26	0.23	0.30	0	0
		1978	7	0.57	0.33	1.12	29	14
		1979	6	0.75	0.47	1.21	67	17
		1980	8	0.54	0.35	0.92	38	0
		1981	34	0.62	0.26	1.26	59	9
		1982	5	0.62	0.57	0.67	100	0
		1983	3	0.49	0.26	0.61	67	0
		1984	1	0.42	0.42	0.42	0	0
		1985	1	0.32	0.32	0.32	0	0
		ALL	85	0.52	0.16	1.26	42	6
	Tullibee	1980	1	0.54	0.54	0.54	0	0
Split 5613 9615	Whitefish	1969	1	0.06	0.06	0.06	0	0
		1970	2	0.09	0.04	0.14	0	0
		ALL	3	0.08	0.04	0.14	0	0
	Pike, N.	1970	5	0.55	0.24	1.09	20	20
		1978	1	0.62	0.62	0.62	100	0
		1979	1	0.74	0.74	0.74	100	0
		1982	1	0.58	0.58	0.58	100	0
		1985	1	0.81	0.81	0.81	100	0
		ALL	9	0.61	0.24	1.09	56	11

* analyses of variance indicates differences in the mean mercury levels between years are significant (P >.95).

LAKE NAME	SPECIES	YEAR	N	MEAN	LOW	HIGH	% REJECTION	
				PPM	PPM	PPM	0.5	1.0
Split 5613 9615	Pickerel*	1970	6	0.76	0.59	0.95	100	0
		1974	5	0.57	0.24	0.77	60	0
		1977	8	0.62	0.47	0.82	88	0
		1978	9	0.59	0.43	0.78	44	0
		1979	1	0.28	0.28	0.28	0	0
		1980	3	0.52	0.35	0.62	67	0
		1981	2	0.87	0.79	0.96	100	0
		1982	1	0.57	0.57	0.57	100	0
		1985	1	0.50	0.50	0.50	0	0
		ALL	36	0.62	0.24	0.96	69	0
	Sturgeon	1977	1	0.13	0.13	0.13	0	0
Stephens 5626 9507	Pickerel	1981	1	1.76	1.76	1.76	100	100
War 5558 9539	Pike, N.	1972	1	0.06	0.06	0.06	0	0
		1975	1	0.25	0.25	0.25	0	0
		ALL	2	0.15	0.06	0.25	0	0
	Pickerel	1971	1	0.16	0.16	0.16	0	0
		1975	1	0.27	0.27	0.27	0	0
		1981	1	0.12	0.12	0.12	0	0
		ALL	3	0.18	0.12	0.27	0	0
Wapisu 5547 9911	Pike, N.	1977	3	1.53	1.48	1.61	100	100
	Pickerel	1977	3	1.33	0.80	1.81	100	67
Wuskwatim 5532 9832	Whitefish	1970	1	0.08	0.08	0.08	0	0
	Pike, N.	1982	1	1.31	1.31	1.31	100	100
	Pickerel	1970	1	0.34	0.34	0.34	0	0
		1971	1	0.25	0.25	0.25	0	0
		1973	1	0.40	0.40	0.40	0	0
		1974	1	0.44	0.44	0.44	0	0
		1975	1	0.35	0.35	0.35	0	0
		1976	1	0.26	0.26	0.26	0	0
		1977	1	0.38	0.38	0.38	0	0
		1983	3	0.81	0.79	0.84	100	0
		1985	2	0.52	0.29	0.76	50	0
		ALL	12	0.49	0.25	0.84	33	0

* analyses of variance indicates differences in the mean mercury levels between years are significant (P >.95).

LAKE NAME	SPECIES	YEAR	N	MEAN PPM	LOW PPM	HIGH PPM	% REJECTION 0.5	1.0
Yawningstone 5421 10051	Whitefish	1973	2	0.04	0.03	0.06	0	0
	Pike, N.	1971	5	0.14	0.06	0.20	0	0
		1973	2	0.23	0.19	0.28	0	0
		1974	1	0.22	0.22	0.22	0	0
		1976	2	0.17	0.17	0.18	0	0
		1981	1	0.22	0.22	0.22	0	0
		ALL	11	0.18	0.06	0.28	0	0
	Pickerel	1971	6	0.24	0.12	0.38	0	0
		1973	2	0.27	0.24	0.30	0	0
		1974	1	0.34	0.34	0.34	0	0
		1976	2	0.22	0.22	0.23	0	0
		1981	1	0.31	0.31	0.31	0	0
		ALL	12	0.26	0.12	0.38	0	0
Yellow Water R. 5747 10132	Pike, N.	1979	1	0.37	0.37	0.37	0	0
		1985	1	0.26	0.26	0.26	0	0
		ALL	2	0.31	0.26	0.37	0	0
	Pickerel	1979	1	0.25	0.25	0.25	0	0
		1985	1	0.18	0.18	0.18	0	0
		ALL	2	0.21	0.18	0.25	0	0

TABLE 2. PRODUCTION vs MERCURY REJECTIONS FOR N. PIKE FROM SELECTED MANITOBA LAKES

LAKE	YEAR	Kg N. PIKE LANDED	 REJECTED
Apussigamasi	1982	438	136
	1983	1163	163
	1984	783	762
Beatty	1984	762	264
Beaucage	1979	225	23
	1982	416	136
	1984	265	111
Drunken	1981	4971	136
Hill	1977	2598	163
Issett	1984	1629	546
Jordan	1978	658	218
Kiskitto	1978	841	163
Laurie	1979	2554	327
McLeod	1983	2075	680
Mynarski West *	1967	85	
	1977	1387	1387
	1982	2269	163
Mystery	1982	2742	408
Nelson R.	1978	1818	1388
	1979	2054	925
	1982	4263	327
	1983	4405	136
	1984	6553	546
Notigi	1978	1425	245
	1982	467	124
Osik	1984	127	27

* All production from this lake, which is on the diversion route, is included. For the other lakes, production is listed only where rejections due to excessive mercury occurred.

TABLE 2 (cont.)

LAKE	YEAR	Kg LANDED	N. PIKE REJECTED
Rat *	1962	702	
	1963	702	
	1964	1080	
	1965	690	
	1966	224	
	1967	7051	
	1969	2211	
	1970	1200	
	1978	4986	4739
Sipiwesk	1978	26423	544
	1979	13834	2123
	1980	27005	544
	1981	39471	5933
	1982	19446	3756
	1983	13886	2096
	1984	30473	327
South Indian	1981	38283	1650
	1982	13826	2313
	1983	12756	600
	1984	18200	517
Stag	1982	1070	191
Wapisu *	1963	214	
	1964	740	
	1966	63	
	1967	518	
	1969	88	
	1977	3449	2068

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* All production from these lakes, which are on the diversion route, is included. For the other lakes, production is listed only where rejections due to excessive mercury occurred.

TABLE 3. PRODUCTION vs MERCURY REJECTIONS FOR PICKEREL FROM SELECTED MANITOBA LAKES

LAKE	YEAR	Kg PICKEREL	
		LANDED	REJECTED
Apussigamasi	1982	266	82
	1983	136	136
Barlow	1978	3152	54
Beatty	1984	220	80
Beaucage	1979	743	102
	1982	1165	218
	1984	496	383
Hill	1977	1857	136
Issett	1984	594	594
Jordan	1978	2693	300
Mynarski West *	1964	429	
	1965	385	
	1966	1688	
	1967	1893	
	1977	3053	
	1980	785	
	1981	2012	
	1982	3382	1034
Notigi	1982	27	15
Rat *	1962	1813	
	1963	1748	
	1964	2566	
	1965	409	
	1966	2742	
	1967	5294	
	1978	5938	2286
Sipiwesk	1981	39421	405
South Indian	1981	25521	218

* All production from this lake which is on the diversion route is included. For the other lakes, production is listed only where rejections due to excessive mercury occurred.

TABLE 3. (continued)

Wapisu *	1963	3188	
	1964	1429	
	1965	121	
	1966	6080	
	1967	760	
	1968	309	
	1969	536	
	1977	4710	2586

---

* All production from this lake, which is on the diversion route, is included. For the other lakes, production is listed only where rejections due to excessive mercury occurred.

TOTAL MERCURY CONCENTRATIONS IN LARGE FISHES FROM LAKES ON THE CHURCHILL RIVER DIVERSION AND NELSON RIVER

A report for the Canada - Manitoba Agreement
on the Study and Monitoring of Mercury in the
Churchill River Diversion
Project 3.3 Mercury in Fish

by

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SPECIAL NOTE

This report was prepared for the Canada - Manitoba Agreement on the Study and Monitoring of Mercury in the Churchill River Diversion. Any conclusions, recommendations, or opinions expressed herein are those of the authors and not necessarily the same as those of the Steering Committee for this Agreement.

EXECUTIVE SUMMARY

Total mercury concentrations in the axial muscle tissue of 13 large fish species from 16 lakes associated with the Churchill River Diversion and 4 lakes on or connected with the Nelson River in northern Manitoba were monitored between 1983 and 1985 under the aegis of the Canada - Manitoba Agreement on the Study and Monitoring of Mercury in the Churchill River Diversion. The primary objective of this monitoring was to determine those species in which the level of mercury contamination presented most concern to the health of residents within the study area who rely on fish as a significant source of food. Three species were found to contain muscle mercury levels which were almost always well in excess of the 0.5 $\mu g.g^{-1}$* Canadian marketing tolerance for mercury in fish. These species were northern pike, walleye and sauger. Some individuals of these species possessed concentrations greater than 4.0 $\mu g.g^{-1}$. The prolonged consumption of pike, walleye and sauger from flooded lakes or stream areas along the Churchill River Diversion is not recommended.

Muscle mercury concentrations in lake whitefish, walleye and northern pike increased as the fish grew larger or older. The degree of mercury contamination in these species was positively related to the extent of flooding on the lake from which the samples were taken. Relatively minor increases (10-20%) in lake surface areas were associated with more than a 100 per cent increase in mercury concentrations in pike and walleye. The inundation of lakes along the Rat - Burntwood river

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* $\mu g.g^{-1}$ = parts per million.

portion of the of the Diversion, which began in 1974, enhanced the mobilization of mercury which in turn caused mercury levels in walleye and pike to increase by as much as five times the background fish mercury level. Background mercury levels in pike and walleye from unimpacted lakes in the area generally ranged between 0.3 and 0.4 $\mu g.g^{-1}$. Highest mercury concentrations were observed in pike and walleye from the Notigi reservoir area where flooding was most severe. No evidence was found to suggest that the level of mercury contamination in fishes from the Churchill River Diversion had significantly declined since its inception.

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INTRODUCTION

Elevated levels of mercury in Manitoba fishes first came to the attention of fisheries investigators in November, 1969. At that time, it was announced that a study by Wobeser et al. (1970) revealed unusually high mercury levels in fish from the South Saskatchewan River immediately downstream of a chloralkali plant in Saskatoon, Saskatchewan. A similar mercury source and fish contamination problem was identified at about the same time in the Wabigoon - English - Winnipeg river system downstream of Dryden, Ontario (Bligh 1970). This information led to the testing and documentation of high fish mercury levels in receiving waters in Manitoba, such as Lake Winnipeg, Cedar Lake and the Saskatchewan River (Derksen 1978). As a consequence of these findings and as a precautionary measure, a ban was imposed on commercial fishing on Lake Winnipeg, Cedar Lake and the Saskatchewan River in April, 1970. Subsequent testing of lakes along the Nelson River system in northern Manitoba indicated the presence of elevated fish mercury levels in Sipiwesk and Split lakes; commercial fishing on these lakes was discontinued in June, 1970 (Manitoba Department of Mines, Resources and Environmental Management, 1970 - 1971 Annual Report).

Mercury sources at Saskatoon and Dryden were attributed primarily to chloralkali plants which utilized the mercury cell process; this process may result in significant losses of elemental mercury or inorganic mercury compounds. Investigators such as Wood et al. (1968), Jensen and Jernelöv (1969) and Abelson (1970) demonstrated that such forms of mercury can be readily methylated in the aquatic environment.

Methylmercury may be rapidly bioaccumulated by fish. In sufficient concentrations the methylmercury in fish is toxic to humans. In 1970, stringent controls were placed on mercury losses from chloralkali plants in Canada. These controls resulted in significant reductions in mercury losses from the above plants to the environment and declines in mercury contamination levels in fish. For example, Armstrong and Scott (1979) observed that the total mercury level in northern pike (Esox lucius) of a specific size from Ball Lake, more than 100 km downstream of Dryden, Ontario, declined by approximately 3 $\mu g.g^{-1}$ over the period 1971 - 1976; declines in mercury concentration were also noted for other fish species. They suggested that these declines were due to the decrease in mercury carried suspended in the water. Although there was a general downstream decrease in mercury contamination levels in fishes in the English - Winnipeg river system, it has never been conclusively demonstrated that mercury emissions at Dryden were directly responsible for the mercury found in Lake Winnipeg fishes. To the contrary, a study on the uptake of mercury by caged rainbow trout (Salmo gairdneri), suspended in the South Saskatchewan River downstream of Saskatoon, indicated that the mercury source in Saskatchewan was not responsible for mercury contamination in Manitoba waters (Uthe et al. 1973). Murray (1978), on the other hand, concluded that upstream industrial and domestic sources were still primarily responsible for the mercury in fish from the Saskatchewan River as far as Cumberland Lake near the Manitoba-Saskatchewan border.

Following the ban on commercial fisheries in Manitoba in 1970, a joint federal - provincial program was initiated to monitor mercury levels in fishes from the affected waters. Information from this monitoring resulted in the re-instatement of the commercial whitefish (Coregonus clupeaformis) fishery in northern Lake Winnipeg during the 1971 summer season and in the re-opening of the entire lake fishery during the following summer. Commercial fishing on Cedar Lake was again permitted during the 1973/74 winter season after a 1973 study (Moshenko and McGregor 1978) had indicated some decline in the quantities of northern pike and walleye (Stizostedion vitreum) rejected because of excessive mercury contamination. The Saskatchewan River in Manitoba was re-opened to commercial fishing in the 1974 summer season, while both Sipiwesk and Split lakes had been fished to a limited extent since 1972.

During the following several years, mercury contamination in excess of the U.S. and Canadian marketing tolerance of 0.5 $\mu g.g^{-1}$ total mercury was observed in predatory fishes such as northern pike, walleyes and occasionally lake trout (Salvelinus namaycush) from approximately 40 northern Manitoba lakes (G. McGregor, personal communication). Although some of the fish with excessive mercury contamination could be exported to European countries which had a higher mercury tolerance, these markets were limited. Approximately two thirds of Manitoba's commercial fish production in still exported to U. S. markets. This placed some economic burden on the commercial fishing industry since fishermen were not paid for rejected fish and a fee was assessed for the cost of mercury testing. In 1979, the U.S. government revised its tolerance for total mercury in fish from 0.5 $\mu g.g^{-1}$ to 1.0 $\mu g.g^{-1}$. As a consequence of this action, fish from most of the above lakes became suitable for export to U.S. markets.

In 1976, Southern Indian Lake on the Churchill River in northern Manitoba was impounded. The mean lake elevation was increased by 3 m and approximately 900 $m^3.s^{-1}$ of water were diverted down the Rat - Burntwood rivers (Churchill River Diversion) for hydroelectric power production on the lower Nelson River (Dickson 1975 and Newbury et al. 1984). Two years earlier, the Notigi Control Dam on the Rat River portion of the Churchill River Diversion was completed. Local runoff into the Rat River resulted in more than a 17 m rise in the Notigi forebay level and extensive shoreline flooding. Following this flooding, increased mercury levels were observed in fish along the Diversion route and in 1977 and 1978 several lakes in the area (Issett, Mynarski, Rat, Notigi and Wapisu) were effectively closed to commercial fishing. These increases in fish mercury levels have been discussed by Bodaly et al. (1984). Post-impoundment increases in fish mercury levels were also witnessed in Southern Indian Lake (Bodaly and Hecky 1979), but they were not large enough to justify curtailment of commercial fishing. Except for a few lakes, the only areas in northern Manitoba that are not presently fished because of excessive mercury concentrations (i.e. >1.0 $\mu g.g^{-1}$) in predatory species are those lakes which are directly influenced or backflooded by the Churchill River Diversion (G. McGregor, personal communication).

The discovery of post - impoundment increases in fish mercury levels in Southern Indian Lake and the Churchill River Diversion led to discussions between environment officials of the federal and provincial governments and the eventual negotiation of the Canada - Manitoba Agreement on the Study and Monitoring of Mercury in the Churchill River Diversion. The study Agreement was formally signed March 10, 1983, but was made retroactive to April, 1982. The Agreement's objectives were to document mercury levels in water, sediment and aquatic food chains, determine sources of mercury and the mechanisms of its release for uptake by aquatic organisms, assess the results in terms of future water management activities, and to inform the public within the study area. The Study Agreement was divided into three major investigative program areas. The first and second programs, respectively, involved studies to determine sources of mercury to the Churchill River Diversion area and to examine pathways of mercury bioaccumulation in the aquatic food chain. Program 3 included studies designed to obtain a more comprehensive understanding of the mercury problem in the Churchill River Diversion and its consequences, particularly from the viewpoint of human health, to residents of the areas. This program encompassed the present study (i.e. Project 3.3 - Mercury in Fish).

The principal objective initially developed for Project 3.3 was to sample several species of fish for mercury analysis from the Churchill River Diversion system, as well as from areas where residents of communities along the Diversion route obtain fish for domestic purposes and from areas not affected by the Diversion. This objective was expanded to include mercury analysis on basically all large fish species encountered in fish sampling activities. Another two added objectives were to document, where possible, long-term trends in the mercury levels of fish from flooded lakes within the study area and to examine species differences in liver mercury: muscle mercury ratio. Since the liver excretes mercury, it was hoped that examination of liver mercury:muscle mercury ratio might reveal short-term trends in mercury uptake by fish. Ancillary to the mercury monitoring objective, information was also coincidently compiled on the principal commercial fish populations of various lakes in the study area. This information has been summarized in a series of data reports by Patalas (1984 a,b), Kirton (1985, 1986 and *in prep*.) and Hagenson (*in prep*.).

DESCRIPTION OF STUDY AREA

The major waters considered in the present study included all lakes, except Issett Lake[1], which were directly or indirectly impacted by the Churchill River Diversion from Southern Indian Lake to Stephens Lake on the lower Nelson River (Fig. 1). Lakes which were directly impacted were those which received diverted flows from the Churchill River, while the indirectly impacted lakes were those that were backflooded to a lesser or greater extent by impoundment and increased flows. The indirectly impacted lakes include West and Central Mynarski lakes which are on the Reading River, a tributary of Rat Lake. These lakes were backflooded by the Notigi Control Dam on the Rat River at the outlet of Notigi Lake. The other indirectly impacted lakes are located downstream of the Notigi Control Dam. Flooding in these areas has been due to the increased diversion flows. Included in these lakes are Macheewin, Osik and Footprint lakes which drain into Threepoint Lake via the Footprint River. Threepoint Lake is formed at the confluence of the Rat, Footprint and Burntwood rivers. The Burntwood River, downstream of Threepoint Lake, flows through Wuskwatim and Apussigamasi lakes. Between these two

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[1] Mercury concentrations in Issett Lake fish are presented by Bodaly and Strange (1986).

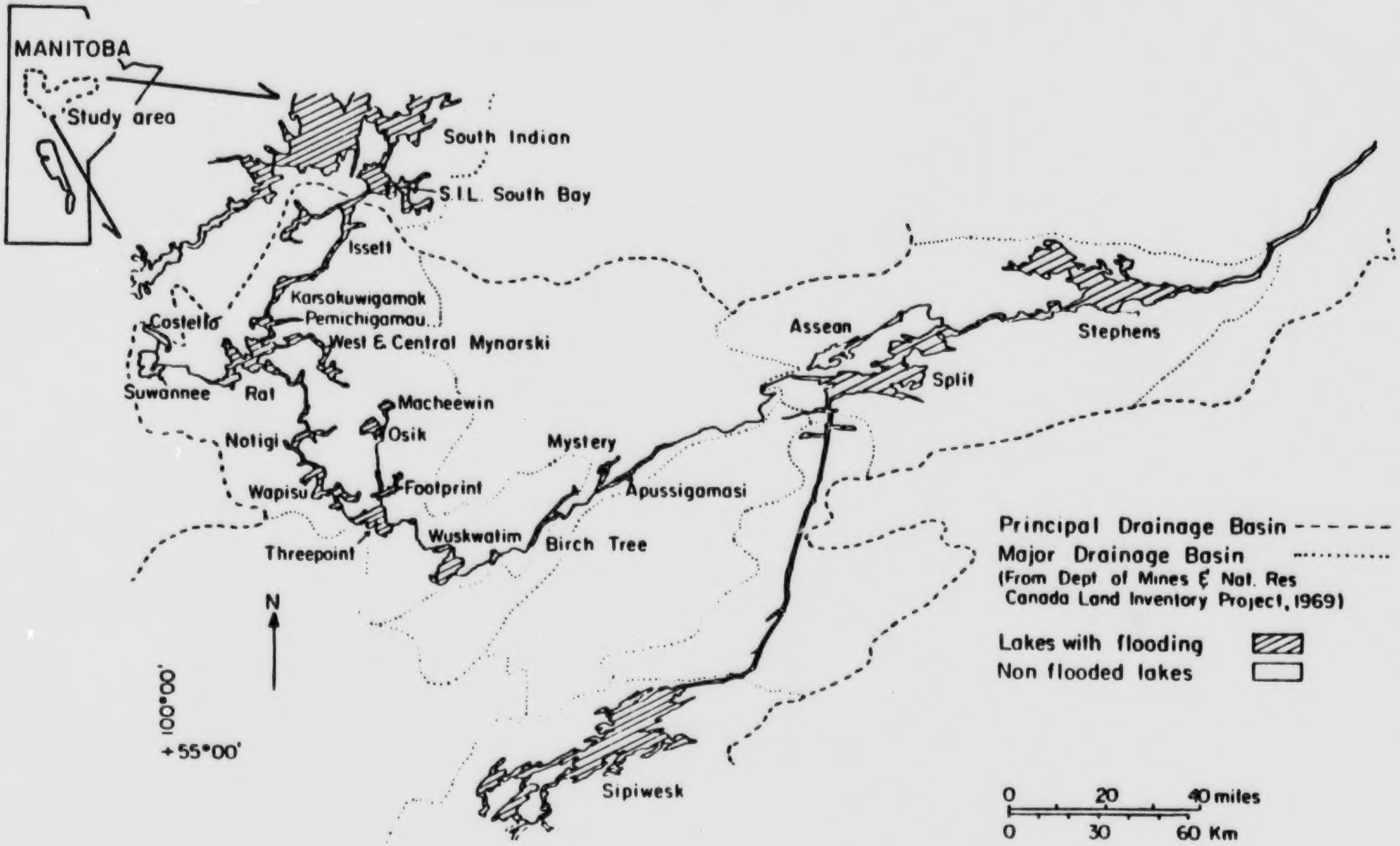
MANITOBA
Study area
South Indian
S.I.L. South Bay
Issett
Karsakuwigamak
Pemichigamau
West & Central Mynarski
Costello
Suwannee
Rat
Macheewin
Osik
Notigi
Wapisu
Footprint
Threepoint
Wuskwatim
Birch Tree
Mystery
Apussigamasi
Assean
Split
Stephens
Sipiwesk
N
100°00'
+55°00'
Principal Drainage Basin
Major Drainage Basin
(From Dept of Mines & Nat. Res
Canada Land Inventory Project, 1969)
Lakes with flooding
Non flooded lakes
0 20 40 miles
0 30 60 Km

Fig. 1. Locations of lakes on the Churchill River Diversion and Nelson river from which fish samples were taken for mercury testing during the period 1983 - 1985. Mercury results for Southern Indian Lake and Issett Lake are presented by Bodaly and Strange (1986).

areas, backflooding has occurred on Birch Tree and Mystery lakes. Some 70 km downstream of Apussigamasi Lake, the Burntwood River joins with the Odei River which then empties into Split Lake on the Nelson River. Although increased flows from the Churchill River Diversion have augmented water levels on Split Lake, there has been no perceptible flooding on the lake[2]. Downstream of Split Lake, the Nelson River widens into Stephens Lake which forms the reservoir of the Kettle Rapids Dam. It is doubtful that the increased diversion flows have caused any additional flooding on Stephens Lake.

In addition to the impacted lakes along the Churchill River Diversion, we considered several other waters. These included Costello and Suwannee lakes, which drain into the west side of Rat Lake via the Suwannee River. Both lakes are elevated well above Rat Lake and have not been impacted by backflooding. Another unimpacted lake sampled was Assean Lake which drains into the east end of Split Lake via the Assean River. Also included in the study area was Sipiwesk Lake on the Nelson River upstream of Split Lake. This lake has been flooded by the Kelsey Dam which is located on the Nelson River just before it enters Split Lake.

All of the above lakes, except Stephens Lake, lie on the

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2 Exceptionally high flows during the present time (July 1986) have resulted in increased lake levels on Split Lake with some flooding along low shorelines.

Precambrian Shield, which consists of low, rolling hills. Most of the area was once covered by Lake Agassiz. With the retreat of this glacial lake, extensive deposits of fine grained glacio-lacustrine material were laid down in the depressions. Rock outcrops throughout the study area are relatively infrequent. Stephens Lake lies on the periphery of the Precambrian Shield and its juncture with the Hudson Bay Lowlands. The Lowlands consist of recently emerged marine deposits. Vegetation in most of the study area consists of northern coniferous forest; the main species is black spruce. Other conifers include white spruce, jack pine and balsam fir. In some places mixed stands occur with the deciduous species chiefly being aspen, white birch and balsam poplar. Stephens Lake lies in a transitional vegetation zone (Weir 1983). In this area the density of stands decreases.

As previously noted, construction of control structures for the Churchill River Diversion was completed in 1976 and effected the diversion of 75 percent of the Churchill River flow to the Nelson River. The Diversion, along with the regulation of outflows from Lake Winnipeg, is employed to augment wintertime flows to hydroelectric generating stations on the lower Nelson River (Dickson 1975). The Diversion and associated areas prior to their development have been described in extensive bio-physical studies conducted between 1971 and 1974 under the Lake Winnipeg, Churchill and Nelson Rivers Study Board (1975). Ecological effects, in terms of both physical and biological changes, of the Churchill River Diversion are outlined in summary by Bodaly et al. (1984). Increased diversion flows from the Churchill River and the Notigi Control Dam have increased the surface elevations of most lakes in the Notigi forebay area by approximately 10m. The greatest water level increase (> 15m) has been on Notigi Lake, immediately above the dam (Newbury et al. 1984). Most of the flooded timber along the main channel in the Notigi reservoir has been removed either by currents and ice action or extracted by work crews. In areas off the main flow channel, such as back bays and coves, the flooded trees which consist mostly of black spruce are still standing and protruding above the water surface. Because of the complex nature of the shoreline and short wind fetches in the Notigi reservoir, shoreline erosion in the area has been minimal.

Flooding along the lower portion of the Churchill River Diversion has not been as severe as it has in the Notigi reservoir. Water levels downstream of the Notigi Control Dam have been increased 3 to 5 m above the historical mean elevations. The increased water levels and river flows have caused significant shoreline erosion and high turbidities. High suspended sediments are especially evident in areas such as Birch Tree Lake, which lies in the middle of deep lacustrine clay deposits and which experienced continued shoreline erosion even before the Diversion came into effect. Water quality conditions in the Rat-Burntwood river system have been altered by the Diversion; these changes are described by Guilbault et al. (1979) and Bodaly et al. (1984).

Few limnological investigations have been conducted on the Nelson River lakes considered in this study. No bio-physical investigations were made on either Sipiwesk Lake prior to its impoundment by the Kelsy Dam in the early 1960's or waters such as Moose Lake which were incorporated into Stephens Lake which formed after the closure of the Kettle Rapids Dam in 1971. Since the studies conducted under the auspices of the Lake Winnipeg, Churchill and Nelson River Study Board (1975), there have been no comprehensive post-development studies on Sipiwesk, Split or Stephens lakes.

METHODS

Fish Mercury Samples

Fish samples for mercury analysis were obtained using standard gangs of multifilament nylon gillnets mainly during openwater, 1983 - 1985. A standard gang consisted of six gillnets of the following mesh sizes: 3.8, 5.1, 7.6, 9.5, 10.8 and 12.7 m (stretched measure). Each gillnet was 22.9 m long and 1.8 m deep. All gillnet sets were made on the bottom and overnight, usually for a duration of 16 - 20 h.

Fork length (total length in case of burbot Lota lota) and round weight were recorded for all fish specimens from which tissue samples were taken for mercury analysis. Sex and state of maturity were recorded and ageing structures were taken from the principal commercial species (i.e. lake whitefish, walleye, sauger Stizostedion canadense, and northern pike). Ageing structures collected from these species were as follows: otoliths and first three pectoral fin rays from whitefish, otoliths and first three dorsal spines from walleye and sauger, and the left cleithrum and first three pectoral fin rays from northern pike. Otoliths, fin rays and spines were thinly sectioned for subsequent age determination. The only preparation required on the cleithra was the removal of any skin or flesh. Fish chosen for mercury testing were selected randomly, but as much as possible, efforts were made to include fish caught in all mesh sizes and from as wide a size range as possible. When catches permitted, the numbers of fish sampled for mercury from each lake each year were 50 per species for whitefish, walleye, sauger and northern pike and 30 specimens of any other large species caught.

Approximately 5 - 10 g of axial muscle and liver tissues were taken from each specimen for mercury testing. Muscle tissue samples were taken consistently in each of the three years. The muscle sample was removed from the area above the lateral line and immediately anterior to the caudal peduncle. Liver tissue samples were taken from all species in 1983, all but suckers (Catostomus and Moxostoma spp.) in 1984, and only whitefish, walleye and northern pike from selected lakes in 1985. The liver sample was generally removed from the posterior tip of the largest lobe. The liver tissue was placed in a separate Whirlpac bag, which was then placed in another Whirlpac bag containing the muscle tissue from the same fish. Date, species, sample number and fish length and weight were noted on the outer bag with a felt-marker pen. Tissue samples were immediately placed on ice and frozen as soon as possible after collection. Samples were submitted to the Freshwater Institute, Canada Department of Fisheries and Oceans for mercury analysis. Mercury testing was done by cold vapour atomic absorption spectro-photometry (Hendzel and Jamieson 1976). All mercury determinations were expressed as $\mu g \cdot g^{-1}$ total mercury (i.e. inorganic and organic mercury) on a wet-weight basis.

Sampling Areas and Flooding Determinations

A total of 16 lakes associated with the Churchill River diversion were sampled for fish mercury in 1983 - 1985 (Table 1). All of these lakes, except two, lie directly on the Diversion route or have been backflooded by the Notigi Control Dam or the increased flows. The two exceptions are Costello and Suwannee lakes which lie on a tributary of Rat

Table 1. Lakes on the Churchill River Diversion and the Nelson River sampled for fish mercury content, 1983 - 1985. Asterisks denote non-flooded lakes.

Lake Name	Geographical Co-ordinates	Years Sampled		
Churchill River Diversion				
Karsakuwigamak	56° 23' 99° 30'	-	1984	-
Pemichigamau	56° 16' 99° 33'	-	-	1985
West & Central Mynarski	56° 10' 99° 12'	-	1984	-
Costello*	56° 13' 100° 09'	-	-	1985
Suwannee*	56° 08' 100° 10'	-	-	1985
Rat	56° 10' 99° 40'	1983	1984	1985
Notigi	55° 57' 99° 18'	1983	-	1985
Wapisu	55° 47' 99° 11'	1983	-	1985
Macheewin	56° 03' 98° 54'	-	1984	-
Osik	55° 58' 98° 58'	-	1984	-
Footprint	55° 47' 98° 53'	1983	1984	1985
Threepoint	55° 41' 98° 56'	1983	1984	1985
Wuskwatim	25° 33' 98° 32'	-	1984	-
Birch Tree	55° 43' 98° 00'	-	-	1985
Mystery	55° 51' 101° 23'	1983	-	-
Apussigamasi	55° 41' 97° 37'	-	1984	-
Nelson River				
Sipiwesk	55° 05' 97° 35'	1983	-	-
Assean*[1]	56° 13' 96° 30'	-	-	1985
Split	56° 08' 96° 15'	1983	1984	-
Stephens	56° 26' 95° 07'	1983	1984	-

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[1] Assean Lake was sampled in winter season 1984/85. All other samples were collected during openwater.

Lake and have not been flooded. Only three lakes, Rat, Footprint and Threepoint, were sampled in all three years of the study. The latter two lakes are situated in the vicinity of Nelson House and provide a source of domestic fish for that community. Four lakes, Sipiwesk, Assean, Split and Stephens, on the Nelson River were also sampled. The latter two are influenced by flows from the Churchill River Diversion. Both Sipiwesk and Split lakes have a reasonably good mercury data record dating back to 1972 (Table 2). The extent of flooding on most of the above inundated lakes was determined from the differences between the pre-and post-impoundment or diversion water surface areas. Water surface areas were measured from 1:50,000 scale topographic maps made prior to and after the development and were calculated using a High State precision coordinate digitizer. Extent of flooding was expressed as the percentage increase in water surface area.

Statistical Methods

For all fish mercury samples collected during the present study, annual determinations were made of the mean length and mean weight, as well as of the mean and range in muscle mercury content of each species sample from each lake. These statistics were also determined for any fish mercury samples collected in years prior to the present study. Detailed statistical analyses were only done for three species: lake whitefish, walleye and northern pike. These species were selected for the following reasons; a) they are important commercial species, b) they are commonly consumed by the local residents in the study area, c) they are, with the exception of ciscoes (Coregonus artedii), the most

Table 2. Years for which information exists on the mercury concentrations in individual whitefish, walleye and northern pike from lakes on the Churchill River Diversion and Sipiwesk Lake. Data in present study were collected in 1983 - 1985.

	Years in which survey samples taken -		
	Whitefish	Walleye	N. pike
W. Mynarski	1984	1982 - 1984	1982 - 1984
Rat	1978, 1980 1983 - 1985	1978 - 1980 1983 - 1985	1978 1983 - 1985
Notigi	1980, 1981 1983, 1985	1978 1980 - 1983 1985	1980 - 1983 1985
Wapisu	1983, 1985	1977 1983, 1985	1977 1983, 1985
Macheewin	1984	1981, 1982 1984	1981 - 1982 1984
Osik	1984	1982 - 1984	1982 - 1984
Footprint	1980, 1981 1983 - 1985	1978 1980, 1981 1983 - 1985	1978 1980, 1981 1983 - 1985
Threepoint	1980, 1981 1983 - 1985	1980, 1981 1983 - 1985	1980, 1981 1983 - 1985
Wuskwatim	1981, 1984	1979 - 1981 1983, 1984	1979 - 1982 1984
Mystery	1983	1979, 1982, 1983	1979, 1982, 1983
Apussigamasi	1984	1982, 1984	1982, 1984
Assean	1981, 1985	1981, 1982 1985	1981, 1982 1985
Split	1983, 1984	1972 - 1974 1976 1978 - 1984	1972, 1973 1976 1978, 1979 1981 - 1984
Stephens	1983, 1984	1983, 1984	1983, 1984
Sipiwesk	1983	1972 - 1974 1976 - 1983	1972 - 1974 1976 - 1983

representative species of the fish communities encountered in the study area and d) much of the fish mercury information collected prior to 1983 was for pike and walleye (see Table 2).

The detailed statistical analyses were as follows. The average fork length, average muscle mercury content and the standard deviation about the mercury mean were determined according to length categories. The length interval used for whitefish and walleye was 50 mm,while a 100 mm interval was used for northern pike. This information was compiled not only for the 1983 - 1985 samples, but also for any previous samples. As will be explained later, this information was used to determine size - standardized mercury concentrations for whitefish, walleye and northern pike.

X - Y plots were made of age and length versus mercury (mercury plotted on the Y - axis) for whitefish, walleye and northern pike. These plots were useful in comparing mercury levels in fish from various lakes and from a given lake over time, particularly in cases where samples had been collected prior to 1983.

Regression analysis was used to examine the relationship of age vs mercury, length vs mercury and weight vs mercury in whitefish, walleye and northern pike taken in the 1983 - 1985 study period. In each case, regressions were calculated using linear, exponential and power functions. Correlation coefficients were also calculated and compared to determine which fish parameter was best correlated with fish mercury content and which mathematical function best described the relationship.

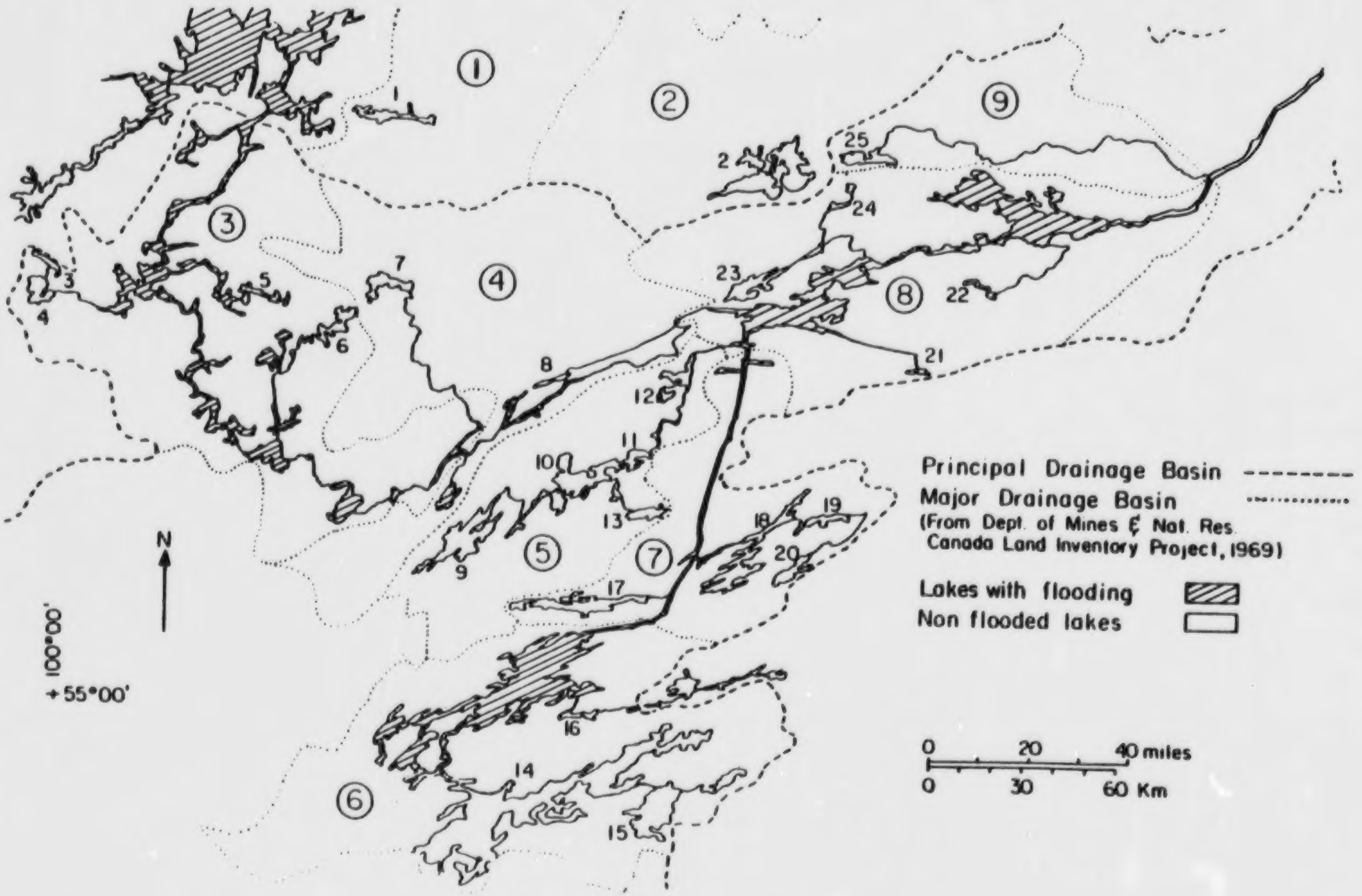
Least squares regressions of age vs mercury and length vs mercury for whitefish, walleye and northern pike were also determined for the data after transformation to $\log_{10}$ values. Where statistically significant ($p \leq 0.05$) correlations were obtained for the 1983 - 1985 data, the regression equations were used to determine size-standardized mercury concentrations for whitefish, walleye and northern pike. Pre-determined lengths for these three species were substituted into the appropriate regression equations to derive the standard mercury concentrations. The standard fork lengths were 350 mm for whitefish, 400 mm for walleye and 550 mm for northern pike. These lengths approximated the modal sizes of these species caught by the 7.6 to 12.7 cm mesh gillnets from all lakes in the study area. The same method was employed by Bodaly and Strange (1986) to derive size-standardized mercury concentrations for fish from Issett Lake and various areas of Southern Indian Lake.

As indicated earlier, another method was used to adjust mercury concentrations of whitefish, walleye and northern pike to a standard length. This method involved the derivation of a coefficient which was then multiplied by the above standard fork lengths. The coefficient for each species from each lake was the quotient of the weighted mean muscle mercury concentration divided by the weighted mean fork length. The mercury levels used to derive the coefficients were taken only from those fish within a specified size category; i.e. 300 - 400 mm for whitefish, 350 - 450 mm for walleye, and 500 - 700 mm for northern pike. This method of standardizing fish mercury concentrations to common sizes was useful in comparing the 1983 - 1985 data with data from other years where sample sizes were frequently inadequate, either in terms of numbers or size range of fish in the sample, for a valid regression analysis.

In cases where sufficient mercury information was available on whitefish, walleye and northern pike for several years or more, the data were examined to determine whether there were any trends in mercury contamination levels through time. Year-to-year comparisons were made of the slopes and elevations (Y - intercepts) of $\log_{10}$ regressions of length vs mercury for each lake using analysis of covariance (Snedecor and Cochran 1967). Comparisons were not made for all possible combinations of years. Rather, the earliest data were compared to the most recent year's data; all subsequent comparisons were generally in sequels of three years. For example, in case the of Notigi Lake whitefish for which data were available in 4 years (1980, 1981, 1983 and 1985), covariance analysis was used to compare the following combinations; 1980 - 1985, 1981 - 1985, 1981 - 1983 and 1983 - 1985. As will be mentioned later, the covariance comparisons did not produce any consistent, meaningful results. Annual comparisons of length vs mercury regressions were therefore attempted by graphically comparing the actual distribution of data points relative to the 95% confidence intervals about the regression lines. These comparisons were only determined for samples from a selected numbers of lakes (i.e. Rat, Notigi, Wapisu, Footprint, Threepoint, Sipiwesk and Split lakes).

The effects of flooding on mercury levels in whitefish, walleye and northern pike from lakes on the Churchill River Diversion and the Nelson River were examined from two approaches. In one approach, the extent of flooding (i.e. percentage increase in lake surface area) was compared to the standardized mercury concentrations in the whitefish, walleye or northern pike by rank correlation. In this case, the standardized mercury concentrations were determined from the regression equations based on the most recent year's data. A second approach was to compare the average and range of size standardized mercury concentrations in whitefish, walleye and northern pike from flooded lakes within a specified drainage basin against similar statistics for fish from non-flooded lakes within the same or adjacent drainage basins shown in Fig. 2. Standardized mercury concentration in these comparisons were derived by the coefficient method described earlier.

The relationships between liver mercury and muscle mercury concentrations in whitefish, walleye and northern pike were examined for various size categories. Although liver tissue samples were collected from a number of lakes, all were not analyzed for mercury. Data are presented here only for Rat, Footprint, Threepoint and Split lakes.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100°00'
+55°00'
Principal Drainage Basin
Major Drainage Basin
(From Dept. of Mines & Nat. Res.
Canada Land Inventory Project, 1969)
Lakes with flooding
Non flooded lakes
0 20 40 miles
0 30 60 Km

Fig. 2. Locations of non-flooded lakes within or adjacent to the study area for which fish muscle mercury data were available. Names of the numbered lakes are presented in Appendix II. Numbers circled refer to the major drainage basins.

RESULTS

Gillnetting in the Churchill River Diversion and Nelson River lakes between 1983 and 1985 produced 15 different fish species and a total of approximately 25,000 individuals. The average catch per standard gang per night was just under 70 fish. The most abundant species caught was cisco, which represented 34.5 per cent of the total catch (Table 3). The next two most abundant species were whitefish (17.4%) and walleye (13.5%). Northern pike and white sucker (Catostomus commersoni) were almost equally represented in the total catch; 10.4 and 10.6 per cent, respectively. Sauger made up 5 per cent of the total catch; much of this catch came from a single area, Wuskwatim Lake. All other species except mooneye (Hiodon tergisus) contributed less than 2 per cent of the total catch.

Mercury concentrations recorded in fish from the study from 1983 to 1985 varied considerably depending on the species and areas sampled as well as on the size or age of the fish. The three species which contained the highest concentrations of mercury were walleye, sauger and northern pike. Samples of these species in almost all cases (>80%) had mean mercury concentrations which exceeded the 0.5 $\mu g \cdot g^{-1}$ Canadian marketing tolerance for mercury in fish (Table 4). These species also possessed the highest individual mercury concentrations found in any of the fishes tested from the study area. Concentrations in excess of 4.0 $\mu g \cdot g^{-1}$ were observed in individual walleye and northern pike, while one sauger contained almost 3.0 $\mu g \cdot g^{-1}$. Mean mercury concentrations in all of the other species were typically less

Table 3. Summary of gillnet catches of fish from the lakes associated with the Churchill River Diversion and several Nelson River lakes, 1983 - 1985.

Species*	No. caught in -			
	1983	1984	1985	Total
Lake whitefish	1,310	1,947	1,046	4,303
Walleye	1,010	1,433	903	3,346
Sauger	393	792	49	1,234
Northern pike	532	1,063	981	2,576
White sucker	506	1,336	782	2,624
Longnose sucker	155	156	23	334
Redhorse sucker**	36	44	8	88
Yellow perch	33	152	257	442
Burbot	48	34	79	161
Freshwater drum	19	1	-	20
Mooneye	523	412	-	935
Goldeye	1	7	1	9
Lake chub	103	68	-	171
Trout-perch	1	1	-	2
Cisco	3,411	4,385	752	8,548
Totals	8,081	11,831	4,881	24,793

* Scientific names given in Appendix I.
** Shorthead redhorse sucker.

Table 4. Distribution of sample means of mercury concentrations in the various fish species collected from the Churchill River Diversion and in the 1983 - 1985 period. The dashed line demarcates the 0.5 $\mu g \cdot g^{-1}$ tolerance for mercury in fish. Numbers in brackets mercury concentrations found in individual fish.

Class Interval (Mercury $\mu g \cdot g^{-1}$)	L. whitefish	Cisco	Walleye	Sauger	Y. perch	N. pike	W. sucker	LN sucker	SH redhorse	Burbot	FW drum
0.00 - 0.09	7	5			2		2	1	1	1	
0.10 - 0.19	4	2	1		5	1	5	3	1	4	1
0.20 - 0.29	15	7	1		5	2	3	5		2	2
0.30 - 0.39	3	10	2		6	1	6	3	1	5	
0.40 - 0.49		4		3	3	2	6	2	3	4	
0.50 - 0.59			2		1	2	1	3		1	
0.60 - 0.69			2	2	1			1			
0.70 - 0.79			3	1			1				
0.80 - 0.89			3			1					
0.90 - 0.99			1								
1.00 - 1.09			3			2					
1.10 - 1.19			4	1		4					
1.20 - 1.29			1	1		3					
1.30 - 1.39			2	1							
1.40 - 1.49			2	2		3					
1.50 - 1.59			3			2					
1.60 - 1.69				5		3					
1.70 - 1.79			3	2		2					
1.80 - 1.89			1	1		1					
1.90 - 1.99						2					
2.00 - 2.09				1							
*											
2.80 - 2.89				1							
	(1.07)	(1.02)	(4.10)	(2.85)	(0.74)	(4.73)	(0.92)	(0.91)	(0.68)	(1.10)	(0.25)

*No sauger were caught with mean mercury concentrations between 2.10 and 2.79 $\mu g \cdot g^{-1}$.

than 0.5 $\mu g.g^{-1}$, although the maximum concentrations in the individuals of some species such as white sucker, longnose sucker (Catostomus catostomus), burbot, whitefish and cisco approached or exceeded the 1.0 $\mu g.g^{-1}$ level. More detailed information on the range in mercury concentrations of individual fish, as well as sample size and average weights and lengths of fish in the 1983 - 1985 and other years' samples are presented in Appendix 4 of Green's (1986) report.

Regression analysis of size/age and mercury data for whitefish, walleye and northern pike indicated that for whitefish and walleye, age generally provided the best correlation with mercury concentrations (Table 5). The correlation between length and mercury in walleye was about as good as that between age and mercury. In the large majority of cases for northern pike, however, length was the best correlate with mercury. Weight was the poorest correlate with mercury in both northern pike and walleye, but in the case of whitefish it was a much better correlate than length.

Regression analysis of the 1983 - 1985 whitefish, walleye and northern pike samples also revealed relatively little difference in the type of mathematical relationship of mercury to size or age between the three species. For walleye and northern pike, the relationship between age/size and mercury content was best described by a power function (Table 6). In contrast, while the length vs mercury relationships for whitefish were usually exponential in nature, linear regression produced the best correlation coefficients for the age vs mercury and weight vs mercury relationships.

Comparisons of length vs mercury regressions for whitefish by

Table 5. Summary of results of regression analysis of age, length and weight vs mercury relationships for whitefish, walleye and northern pike from Churchill River Diversion and Nelson River lakes, 1983 - 1985. Numbers presented under the age, length and weight columns indicate the number of times the regression between that variable and mercury content yielded the highest regression coefficient (r^2).

		No. of times variable vs mercury produced highest r^2**								
		Whitefish			Walleye			N. pike		
Year	No. of Correlations*	Age	Length	Weight	Age	Length	Weight	Age	Length	Weight
1983	7 x 3	13	1	7	12	9	0	4	17	0
1984	11 x 3	22	3	8	21	12	0	8	25	0
1985	9 x 3	23	2	2	16	9	2	4	21	2
Total	27 x 3	58	6	17	49	30	2	16	63	2

* Number of lakes x number of regression functions (i.e. linear, exponential & power).

** r^2 values presented by Green (1986) - Appendix 7.

Table 6. The mathematical functions best describing the relationships of age, length and weight of whitefish, walleye and northern pike to mercury, based on data compiled from the Churchill River Diversion and Nelson River lakes, 1983 - 1985.

	Whitefish			Walleye			N. pike		
	Age	Length	Weight	Age	Length	Weight	Age	Length	Weight
1983	linear	expon.	linear	power	expon. power	power	power	power	power
1984	linear	expon.	expon.	power	expon.	power	power	power	power
1985	linear	expon.	linear	power	power	power	power	power	power

analysis of covariance were in most cases statistically non-significant (Table 7). Where significant differences ($P \leq 0.05$) were found, they were generally in terms of elevation. Significant differences in elevation were recorded in 8 (42%) of 19 comparisons. Only 4 comparisons (21%) exhibited a significant difference in slope; 3 of these comparisons also showed a significant difference in elevation. These coinciding slope and elevation differences were observed in the Notigi Lake whitefish samples (Table 7). Scrutiny of the actual data for Notigi Lake whitefish indicates that the difference recorded for the 1980 - 1985 comparison was obviously due to the poor sample size in 1980 (Fig. 3). Comparison of the 1981 whitefish samples to those of 1983 and 1985 also indicated significant slope and elevation differences. However, considering mercury concentrations of individual whitefish of similar size (e.g. 300 mm), graphic comparisons revealed approximately the same distributional patterns in 1981 and 1983, but mercury concentrations appeared to be higher in 1985 (Fig. 3). Covariance analysis indicated a significant difference in elevation for the 1983 - 1985 comparison. Whitefish in the both 1983 and 1985 samples fell within approximately the same size range, 150 - 500 mm. Examination of the age-mercury distribution, however, indicated that the whitefish in the 1985 sample were generally 2 to 3 years older than those in the 1983 sample. This situation, which underlines the importance of considering age, as well as the size range of fish and sample size, is also apparent for the 1983 - 1985 samples of whitefish from Wapisu Lake (Fig. 4).

Results of the covariance analysis on walleye were similar to those on whitefish; i.e. significant differences were more frequently found in the elevation than in the slope of regression lines. Significant

Table 7. Results of covariance analysis on whitefish from the Churchill River Diversion and Nelson River lakes.

Lake	Years of comparison	Slope		Elevation	
		d.f.	F	d.f.	F
Rat	1978-1985	1,54	0.0728	1,55	3.1911
	1978-1980	1,25	0.2233	1,26	0.8640
	1980-1983	1,43	1.2500	1,44	0.9548
	1983-1985	1,72	6.7740*	1,73	0.2694
Notigi	1980-1985	1,52	6.2807*	1,53	44.7645*
	1981-1985	1,84	6.3292*	1,85	5.4936*
	1981-1983	1,84	6.9954*	1,85	7.1524*
	1983-1985	1,96	0.2056	1,97	25.0073*
Wapisu	1983-1985	1,76	0.0540	1,77	10,4292*
Footprint	1980-1985	1,53	0.1839	1,54	4.3529*
	1981-1985	1,71	0.0681	1,72	4.4470*
	1981-1983	1,43	0.2784	1,44	2.3977
	1983-1985	1,74	1.0835	1,75	0.5492
Threepoint	1980-1985	1,54	1.6268	1,55	1.8670
	1981-1985	1,65	0.6182	1,66	0.3266
	1981-1983	1,37	1.6824	1,38	1.2900
	1983-1985	1,74	0.5474	1,75	0.6436
Split	1983-1984	1,53	1.9800	1,54	4.6184*
Stephens	1983-1984	1,56	1.6531	1,57	0.5541

* significantly different $P \leq 0.05$

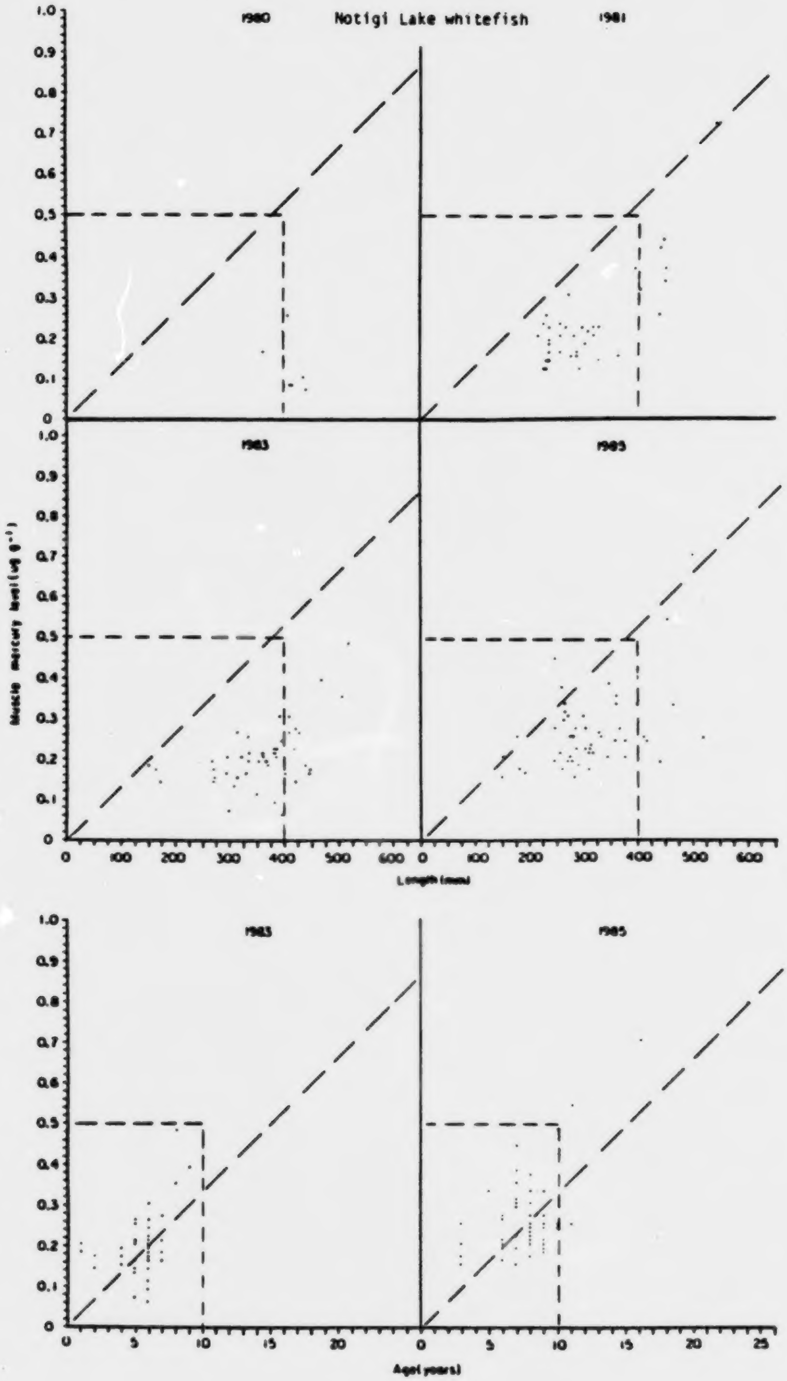

Fig. 3. Plots of whitefish mercury data for Notigi Lake in 1980, 1981, 1983 and 1985. Rectangles, and 45° lines on the graphs are provided to facilitate cross-reference between panels.

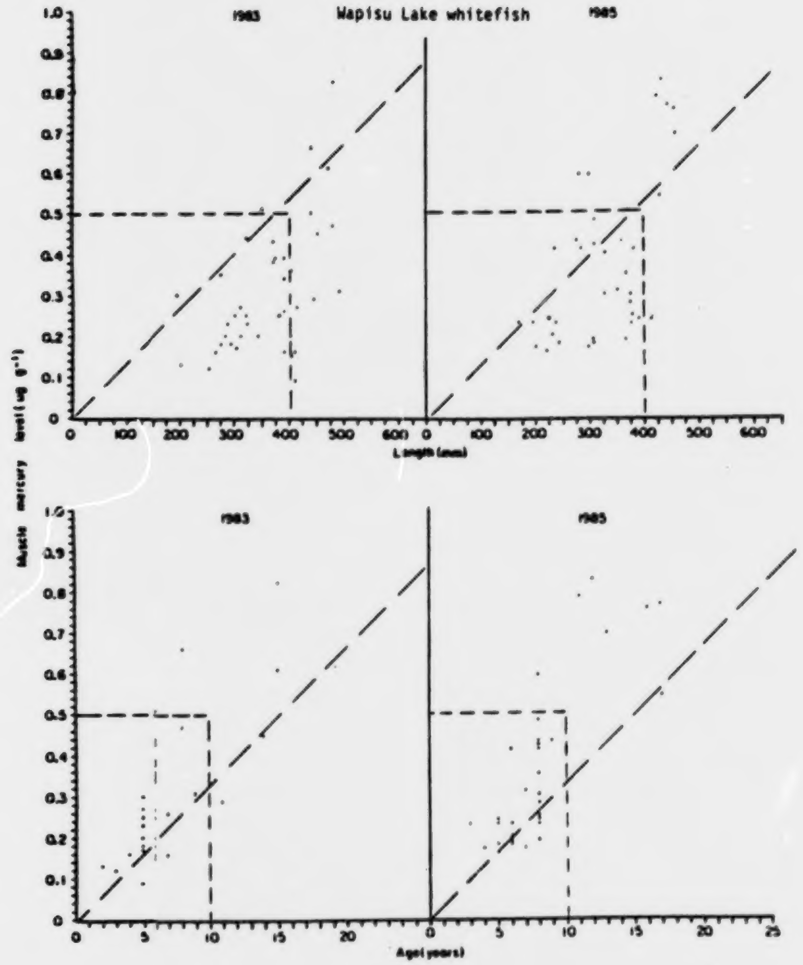

Fig. 4. Plots of whitefish mercury data for Wapisu Lake, 1983 and 1985. Rectangles and 45° lines are provided to facilitate cross-reference between panels.

differences in elevation were noted in 27 (61%) instances, while the slope differed significantly in only 12 (27%) of the 44 comparisons (Table 8). Both slope and elevation differences were noted for 8 comparisons; these were recorded for walleye from Rat, Notigi, Macheewin, Threepoint, Sipiwesk and Split lakes. With reference to Rat Lake walleye , the difference found in elevation between the 1980 and 1983 regression lines was due to the poor representation of individuals less than 400 mm, as well as larger specimens, in the 1980 sample (Fig. 5). The plots for the 1983 and 1984 samples were very similar, since mercury concentrations in walleye less than and greater than 400 mm showed much the same distributions (Fig. 5). The distribution pattern for the 1985 sample data, however, differed from the patterns obtained for the 1983 and 1984 samples; i.e. mercury concentrations were lower in the larger walleye in the 1985 sample. Consideration of the age vs mercury plots for the 1984 - 1985 samples (Fig. 5) indicates a general downward shift in the mercury levels in walleye older than 15 years. This would suggest that a slight drop in the level of mercury contamination was experienced between 1983 and 1985. This downward shift was also apparent among walleye sampled from Notigi lakes in 1983 and 1985 (Fig. 6). Analysis of covariance indicated a significant difference in elevation between regression lines determined for the 1983 and 1985 Notigi Lake walleye samples (Table 8). Similar circumstances to those noted for Rat Lake and Notigi Lake walleye were observed for walleye samples from Threepoint Lake between 1981 and 1985 (Fig. 7).

As in whitefish and walleye, significant differences were most often observed in the comparisons of regression line elevations for northern pike. Significant differences in elevations were observed in 26

Table 8. Results of covariance analysis on walleye from Churchill River Diversion and Nelson River lakes.

Lake	Years of comparison	Slope		Elevation	
		d.f.	F	d.f.	F
West & Central Mynarski	1982-1984	1,78	2.7414	1,79	0.8179
	1982-1983	1,43	1.0445	1,44	9.4631*
	1983-1984	1,59	0.4733	1,60	20.1208*
Rat	1978-1985	1,76	3.0672	1,77	40.6342*
	1978-1980	1,44	5.9683*	1,45	75.0821*
	1980-1983	1,68	0.5623	1,69	75.2472*
	1983-1985	1,100	12.5724*	1,101	8.8044*
Notigi	1978-1985	1,40	0.1532	1,41	2.5237
	1978-1981	1,44	0.3733	1,45	13,0639*
	1981-1983	1,51	5.5999*	1,52	9.6857*
	1983-1985	1,47	2.1560	1,48	4.2630*
Wapisu	1977-1985	1,151	0.0075	1,152	35.3254*
	1977-1983	1,137	0.6817	1,138	9.5816*
Macheewin	1981-1984	1,46	6.6955*	1,47	18.7108*
	1981-1982	1,8	5.1356	1,9	2.1440
	1982-1984	1,46	0.7049	1,47	34.9531*
Osik	1982-1984	1,53	0.8671	1,54	0.7445
	1982-1983	1,14	0.1028	1,15	4.1372
	1983-1984	1,59	1.7379	1,60	1.9165
Footprint	1978-1985	1,73	0.3710	1,74	5.7592*
	1978-1981	1.46	0.0816	1,47	9.3486*
	1981-1983	1,59	1.9670	1,60	1.0158
	1983-1985	1,86	0.1284	1,87	1.8287
Threepoint	1981-1985	1,97	5.8594*	1,98	13.0626*
	1981-1983	1,88	6.6073*	1,89	7.2276*
	1983-1985	1,105	0.1602	1,106	40.1379*
Wuskwatim	1979-1984	1,110	2.4425	1,111	15,6009*
	1979-1981	1,87	0.9860	1,88	0.0040
	1981-1984	1,87	3.4491	1,88	15.1383*
Apussigamasi	1982-1984	1,55	0.0632	1,56	0.4186
Sipiwesk	1972-1983	1,106	21.6372*	1,107	2.8491
	1972-1974	1,72	1.6866	1,73	11.2126*
	1974-1976	1,83	0.8712	1,84	19.5617*
	1976-1978	1,108	0.1079	1,109	2.0141
	1978-1980	1,88	9.3811*	1,89	0.3669
	1980-1983	1,97	55.5786*	1,98	5.5863*
Split	1972-1984	1,94	6.4456*	1,95	0.00005
	1972-1974	1,45	1.1320	1,46	0.0376
	1974-1976	1,62	0.6943	1,63	8.0113*
	1976-1978	1,109	0.8630	1,110	3.4706
	1978-1980	1,77	5.7178*	1,78	2.6655
	1980-1982	1,24	7.8768*	1,25	26.8117*
	1982-1984	1,77	2.4567	1,78	24.7445*
Stephens	1983-1984	1,70	0.0644	1,71	24.0198*

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* significantly different $P \leq 0.05$

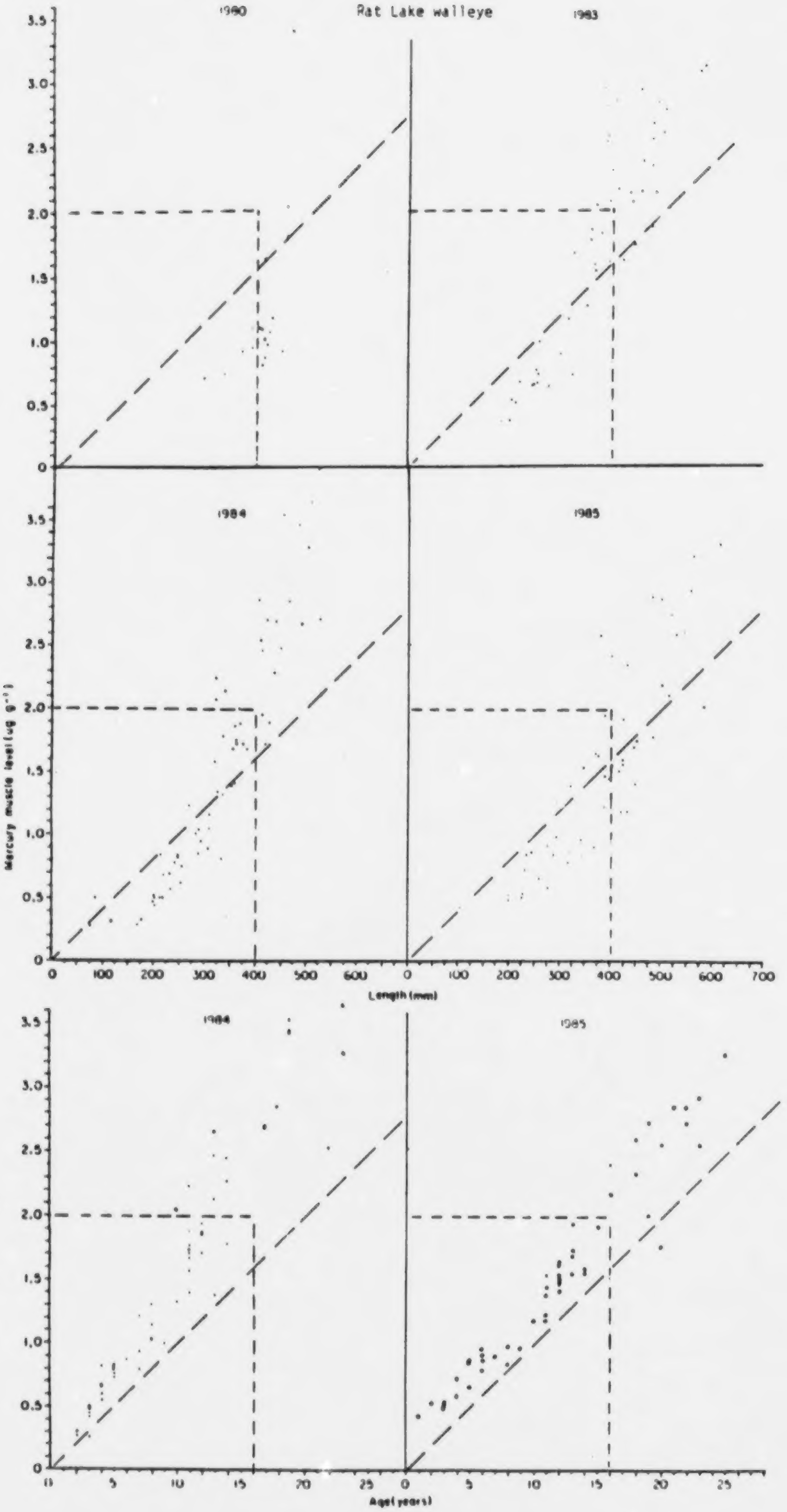

Fig. 5. Plots of walleye mercury data for Rat Lake, 1980 and 1983 - 1985. Rectangles and 45° lines are provided to facilitate cross-reference between panels.

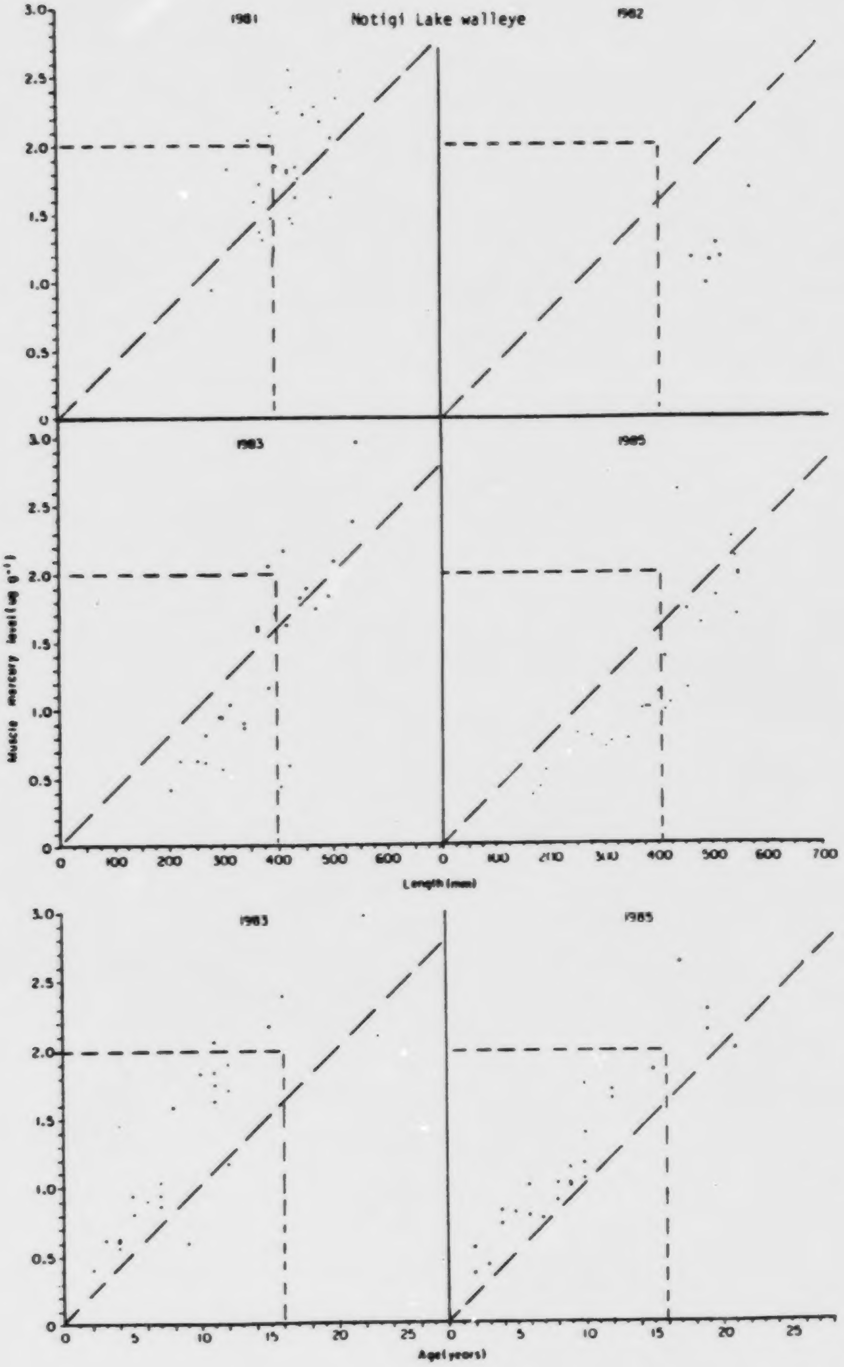

Fig. 6. Plots of walleye mercury data for Notigi Lake, 1981 - 1983 and 1985. Rectangles and 45° lines are provided to facilitate cross-reference between panels.

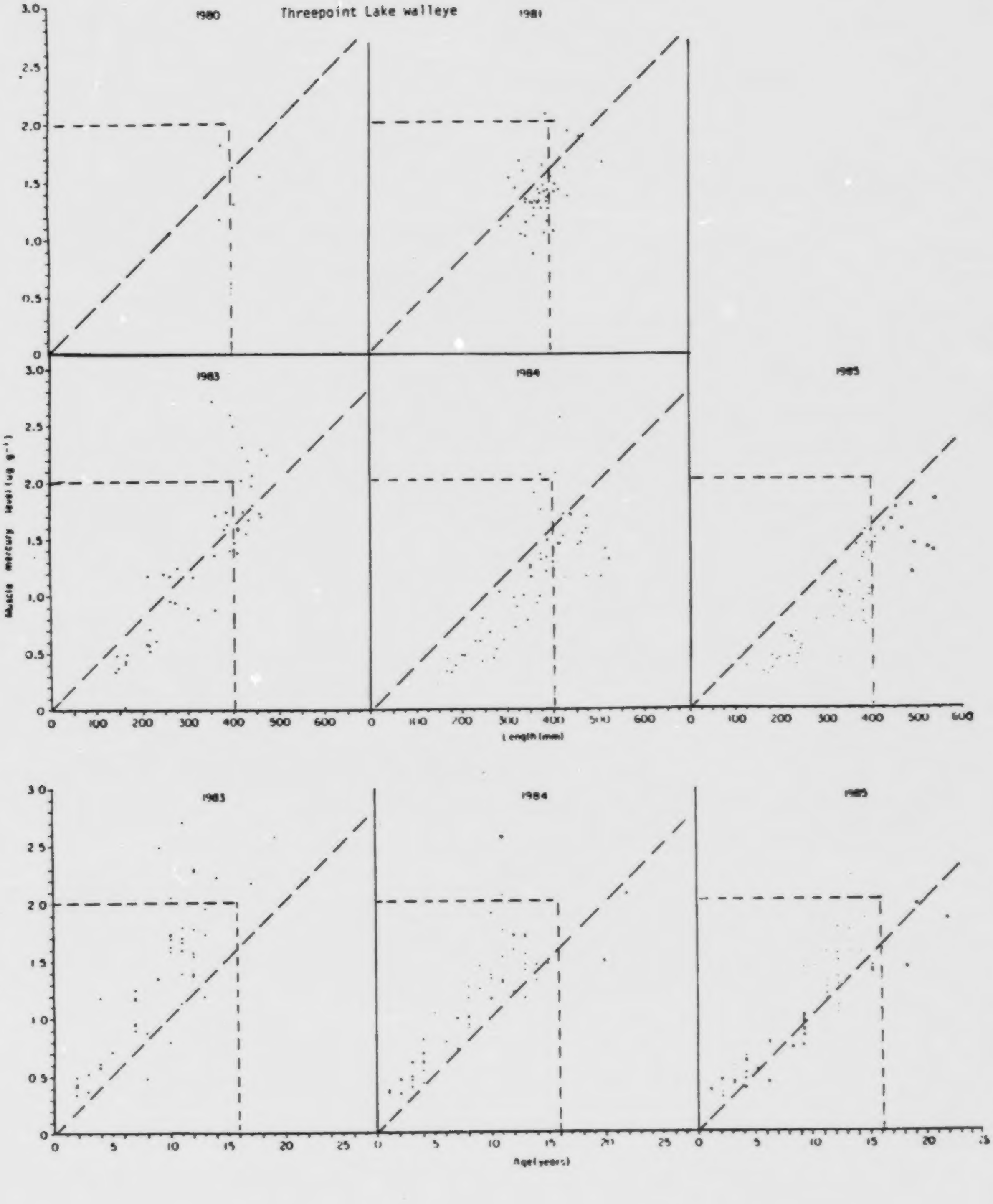

Fig. 7. Plots of walleye mercury data for Threepoint Lake, 1980, 1981 and 1983 - 1985. Rectangles and 45° lines are provided to facilitate cross-reference between panels.

(59%) cases, whereas only 9 (20%) of the 44 comparisons showed significant differences in slope (Table 9). Six of the 9 comparisons showing a significant difference in slope also differed in elevations; these were observed for pike samples from Mynarski, Rat, Wuskwatim, Sipiwesk and Split lakes. Significant differences were observed in the slope and elevation of regression lines for the 1978 and 1985 pike samples from Rat Lake (Table 9). Close examination of the data indicates these differences were probably the result of the poor representation of smaller pike ($\leq$ 500 mm) in the 1978 sample. The 1978 sample regression also differed from that of the 1983 sample in terms of slope. However, there was no marked difference in either slope or elevation between the 1983 and 1985 Rat Lake pike samples. Although the 1983 sample was not as large and contained fewer small fish than the 1985 sample, distributions of the data points for both samples were very similar (Fig. 8). There did appear to be some difference between the 1984 and 1985 samples, although this was not tested statistically. Mercury concentrations in most Rat Lake pike 600 mm and longer tended to be higher in 1984 than in 1985; this difference was also reflected in the age vs mercury plots (Fig. 8). From this, it would seem that mercury levels in Rat Lake pike declined between 1984 and 1985. In contrast to this decline, mercury concentrations in Wuskwatim Lake pike appear to have increased between 1979 - 1981 and 1984 (Fig. 9). Analysis of covariance indicated significant differences between the 1979 and 1981 samples and between the 1981 and 1984 samples.

Large samples of pike were taken from Sipiwesk and Split lakes in a number of years since 1972. For both lakes, significant differences were found in the slopes and elevations of regression lines determined

Table. 9. Results of covariance analysis on northern pike from Churchill River Diversion and Nelson River lakes.

Lake	Years of comparison	Slope		Elevation	
		d.f.	F	d.f.	F
West & Central Mynarski	1982-1984	1,59	34.1430*	1,60	9.7913*
	1982-1983	1,13	0.7873	1,14	0.1985
	1983-1984	1,52	2.9862	1,53	67.5764*
Rat	1978-1985	1,78	21.2876*	1,79	20.495*
	1978-1983	1,42	1.6382	1,43	10.3311*
	1983-1985	1,76	1.7871	1,77	0.3140
Notigi	1980-1985	1,56	1.2877	1,57	0.5124
	1980-1981	1,51	1.1958	1,52	6.8007*
	1981-1983	1,90	1.4358	1,91	11.2405*
	1983-1985	1,95	8.5300*	1,96	2.4918
Wapisu	1977-1985	1,75	1.1314	1,76	60.2601*
	1977-1983	1,87	3.1204	1,88	65.5820*
	1983-1985	1,90	5,2730*	1,91	0.2607
Macheewin	1981-1984	1,58	1.1938	1,59	21.3330*
	1981-1982	1,11	1.9621	1,12	9.5892*
	1982-1984	1,63	1.6239	1,64	0.4768
Osik	1982-1984	1,61	0.5365	1,62	10.0903*
	1982-1983	1,6	0.0002	1,7	0.2001
	1983-1984	1,61	0.2989	1,62	3.6941
Footprint	1978-1985	1,62	2.4687	1,63	8.9590*
	1978-1981	1,28	0.9407	1,29	30.5528*
	1981-1983	1,51	0.2549	1,52	1.7289
	1983-1985	1,85	3.5263	1,86	8.8452*
Threepoint	1981-1985	1,73	0.0004	1,74	1.4341
	1981-1983	1,74	0.4248	1,75	8.5963*
	1983-1985	1,95	2.6519	1,96	19.9626*
Wuskwatim	1979-1984	1,89	29.9887*	1,90	23.0318*
	1979-1981	1,76	2.6750	1,77	0.0289
	1981-1984	1,59	2.3035	1,60	24.4152*
Apussigamasi	1982-1984	1,60	0.0681	1,61	23.4966*
Sipiwesk	1972-1983	1,194	7.1279*	1,195	13.9780*
	1972-1974	1,170	0.0920	1,171	1.2537
	1974-1976	1,172	0.9539	1,173	4.2328*
	1976-1979	1,183	0.0546	1,184	0.8206
	1979-1981	1,105	23.9838*	1,106	0.7554
	1981-1983	1,136	4.6539*	1,137	8.0930*
Split	1972-1984	1,82	6.3652*	1,83	126.9511*
	1972-1973	1,93	0.2029	1,94	2.8912
	1973-1976	1,130	1.5766	1,131	1.6306
	1976-1978	1,102	0.0094	1,103	3.5519
	1978-1981	1,26	0.0613	1,27	8.5183*
	1981-1982	1,40	1.9149	1,41	2.5234
	1982-1984	1,71	0.4300	1,72	33.5970*
Stephens	1983-1984	1,85	0.0552	1,86	8.8533*

* significantly different $P \leq 0.05$

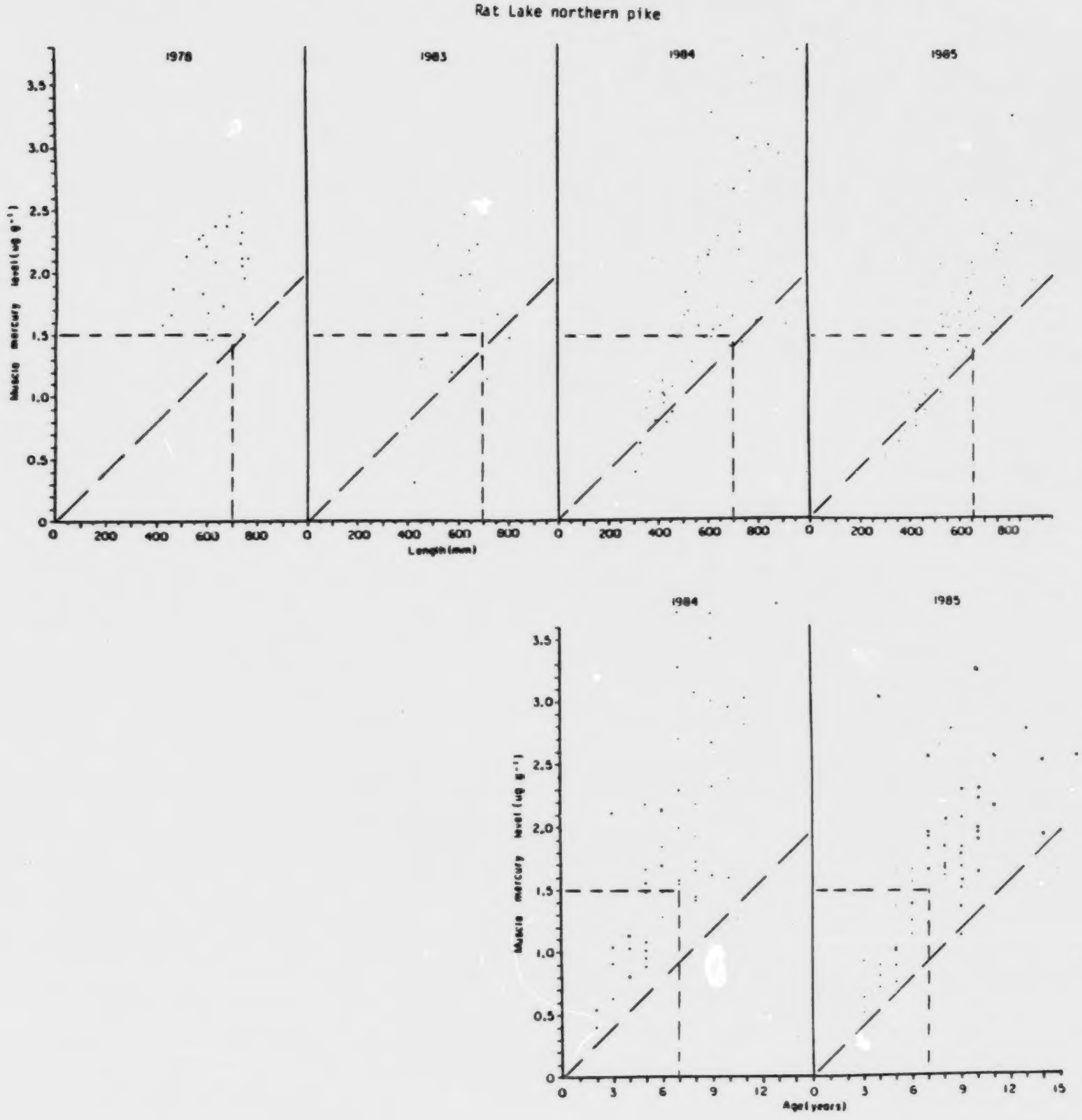

Fig. 8. Plots of northern pike mercury data for Rat Lake, 1978 and 1983 - 1985. Rectangles and 45° lines are provided to facilitate cross reference between panels.

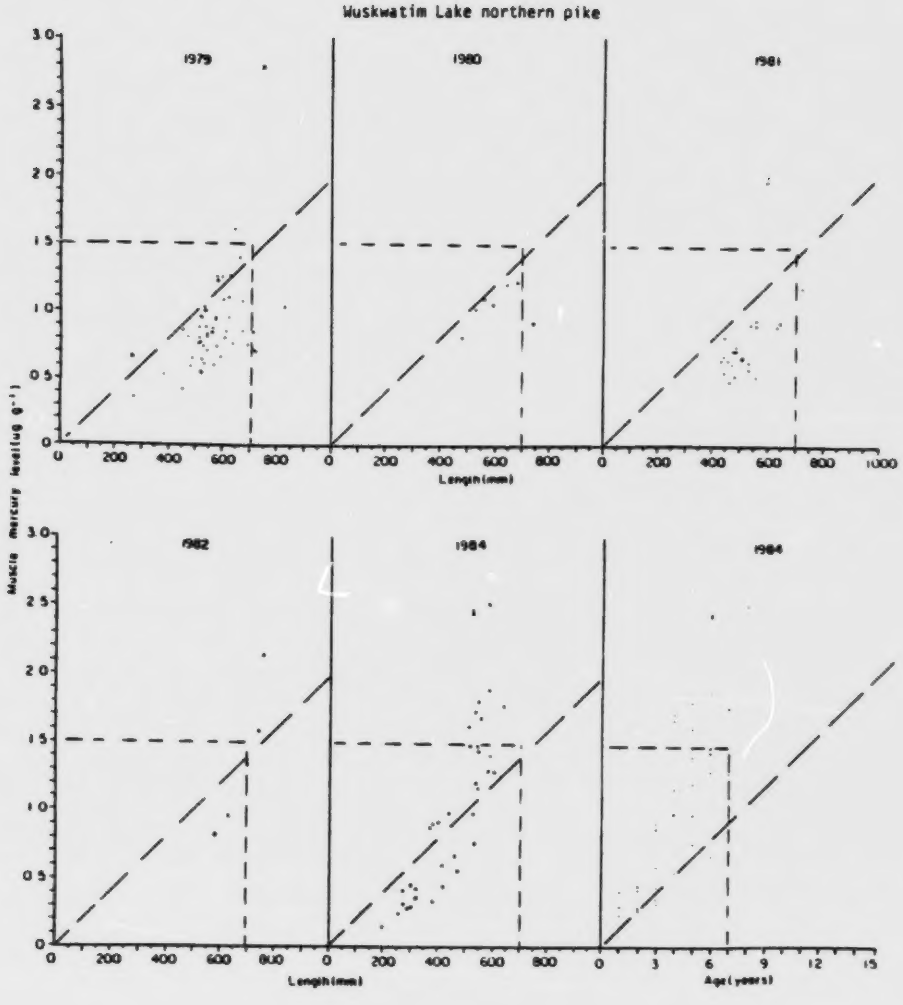

Fig. 9. Plots of northern pike mercury data for Wuskwatim Lake, 1979 - 1982 and 1984. Rectangles and 45° lines are provided to facilitate cross-reference between panels.

for the 1972 and most recent samples (Table 9). Graphic presentation of the data for Sipiwesk Lake and Split Lake pike (Fig. 10 & 11), however, again illustrate the influence that both the size range and number of specimens in the samples may have on the regression results.

The above indicates that regression results for a number of fish mercury samples collected prior to, and in some cases during the present study were not valid for making annual comparisons in the level of mercury contamination in a given species. The validity of such samples was questionable mainly because of the number and size range of specimens represented in the samples. In some cases, the variability in the age -length relationship of fish in the samples was a factor which influenced the regression results. Because age information was only available for the 1983 - 1985 samples, this influence could not be fully examined. Another two factors that may have introduced some further annual variations to the size vs mercury regressions are the exact locations and times the samples were taken. An attempt to determine whether there were localized differences in fish mercury concentrations within a lake indicated no such variations. However, this test was not very sensitive since only a maximum of 50 fish of a species were sampled in any given year. When this sample was divided into sub-lake areas, sample sizes were usually too small to allow valid comparisons. It should also be noted that the regressions for the fish mercury samples collected during this study generally did not exhibit as much variability as those taken before 1983. This may have been due to the fact that samples collected from 1983 to 1985 were taken by basically the same people using the same sampling methods and fishing approximately the same areas in each lake at the same time each year.

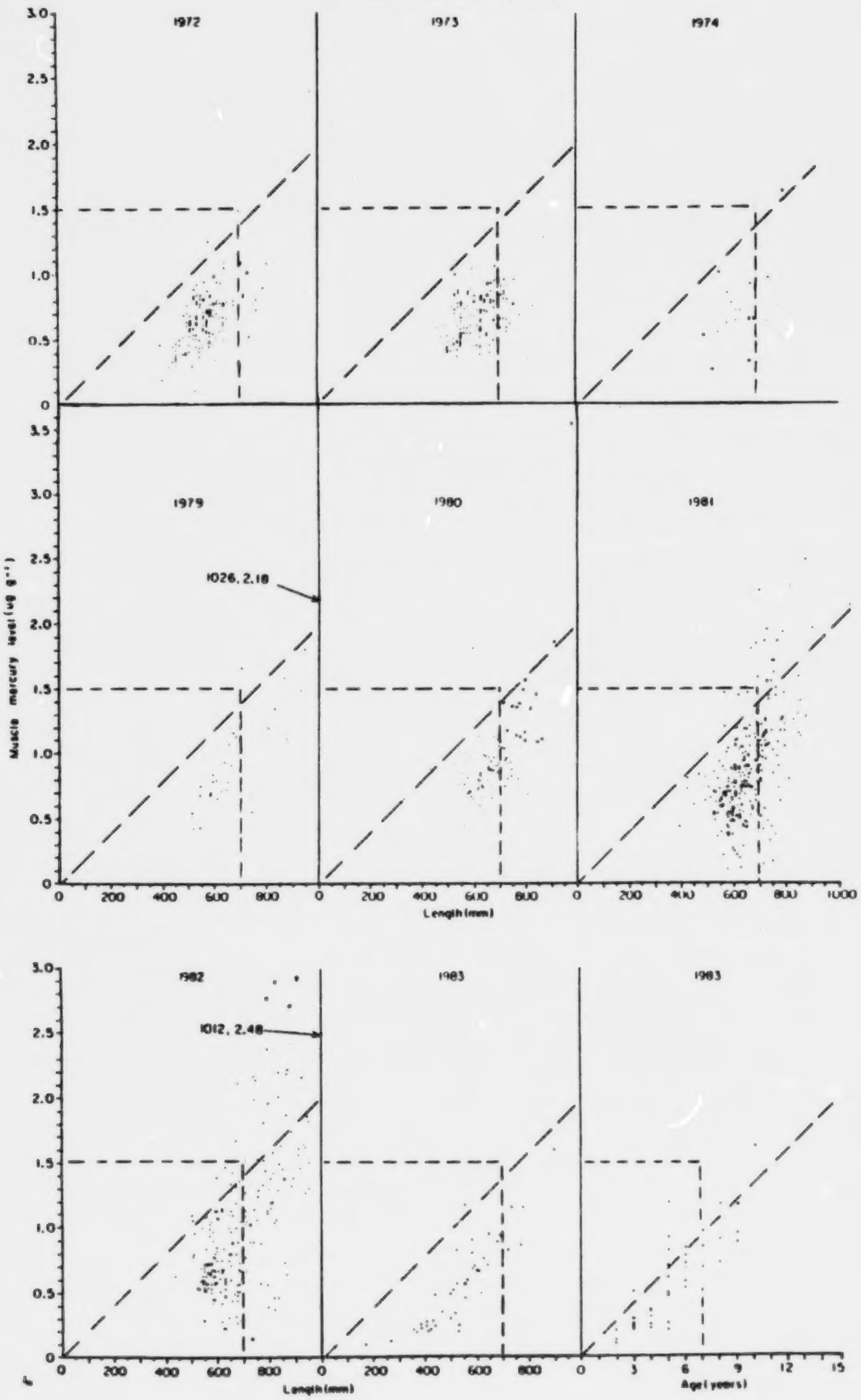

Fig. 10. Plots of northern pike mercury data for Sipiwesk Lake, 1972 - 1974, and 1979 - 1983. Rectangles and 45° lines are provided to facilitate cross-reference between panels.

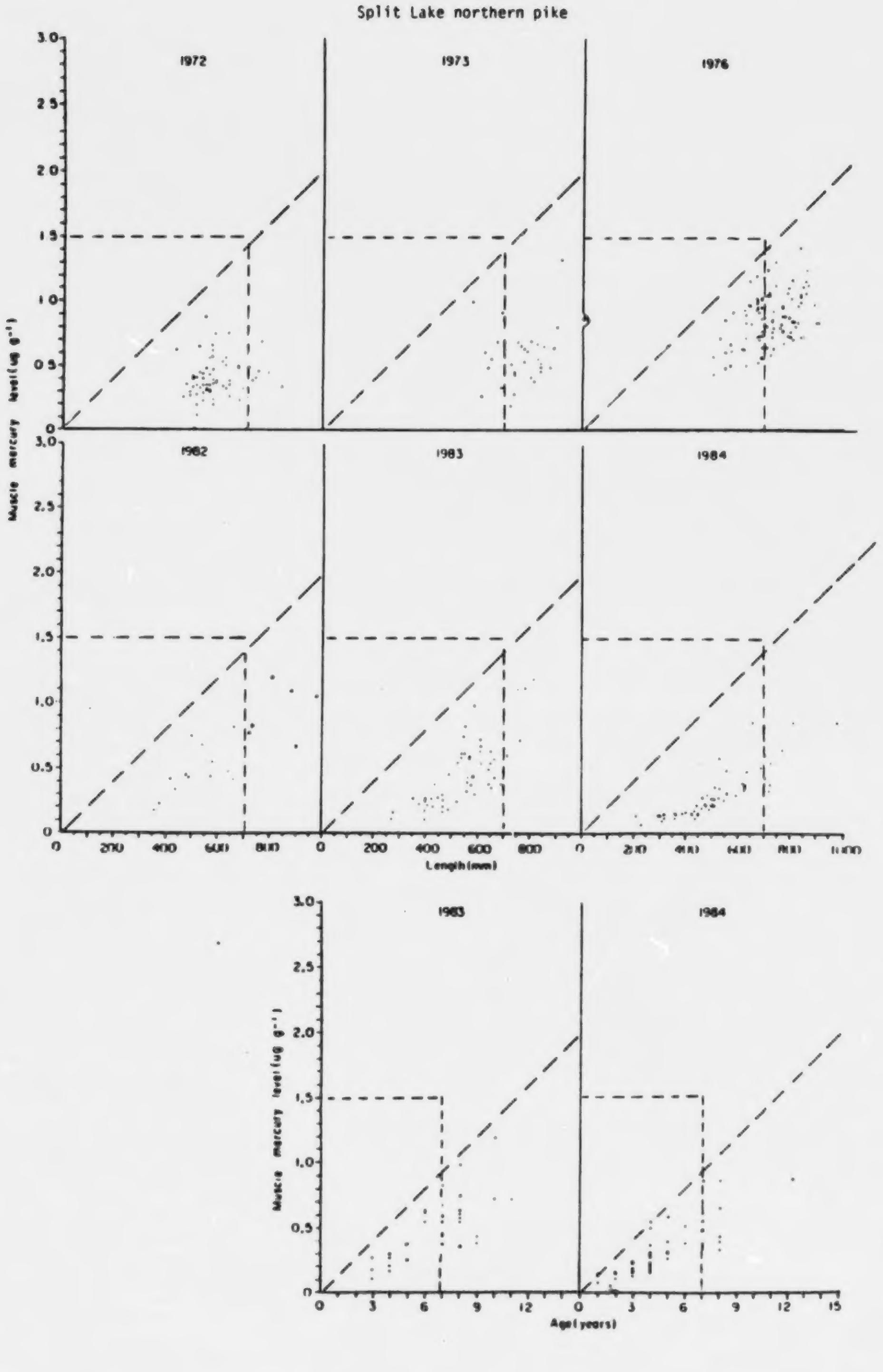

Fig. 11. Plots of northern pike mercury data for Split Lake, 1972, 1973, 1976 and 1982 - 1984. Rectangles and 45° lines are provided to facilitate cross-reference between panels.

In order to ascertain whether there were any temporal changes or trends in the level of mercury contamination of a given species within a lake, the graphic distribution of data points for samples collected prior to the present study were compared to the $\log_{10}$ regression lines for the 1983 - 1985 samples and their 95% confidence intervals for the prediction of individual Y values. If the earlier sample data fell within these confidence intervals for the 1983 - 1985 samples, it was concluded that the early and recent samples were essentially from the same population; i.e. there was no difference in the levels of contamination and that any differences were due to sampling variations. The following text presents the results for such comparisons for whitefish, walleye and northern pike from selected lakes. Particular emphasis is given to lakes which have several or more years of fish mercury information.

Pre - 1983 whitefish mercury data were only available for lakes along the Churchill River Diversion, but they were not collected prior to the completion of the Diversion. Most of the whitefish samples collected before 1983 contained small numbers of fish within a restricted size range. Comparison of sample data and the corresponding regression lines and confidence intervals for Rat Lake whitefish are presented in Fig. 12. Data points for the 1978, 1980 and 1983 samples, which were based on large individuals from a relatively limited size range, typically fell within the 95% confidence envelope for both the 1984 and 1985 regression lines, which were based on fish of a broader size range. The greater representation of individual sizes in the latter samples had the effect of increasing the confidence intervals about the regression lines.

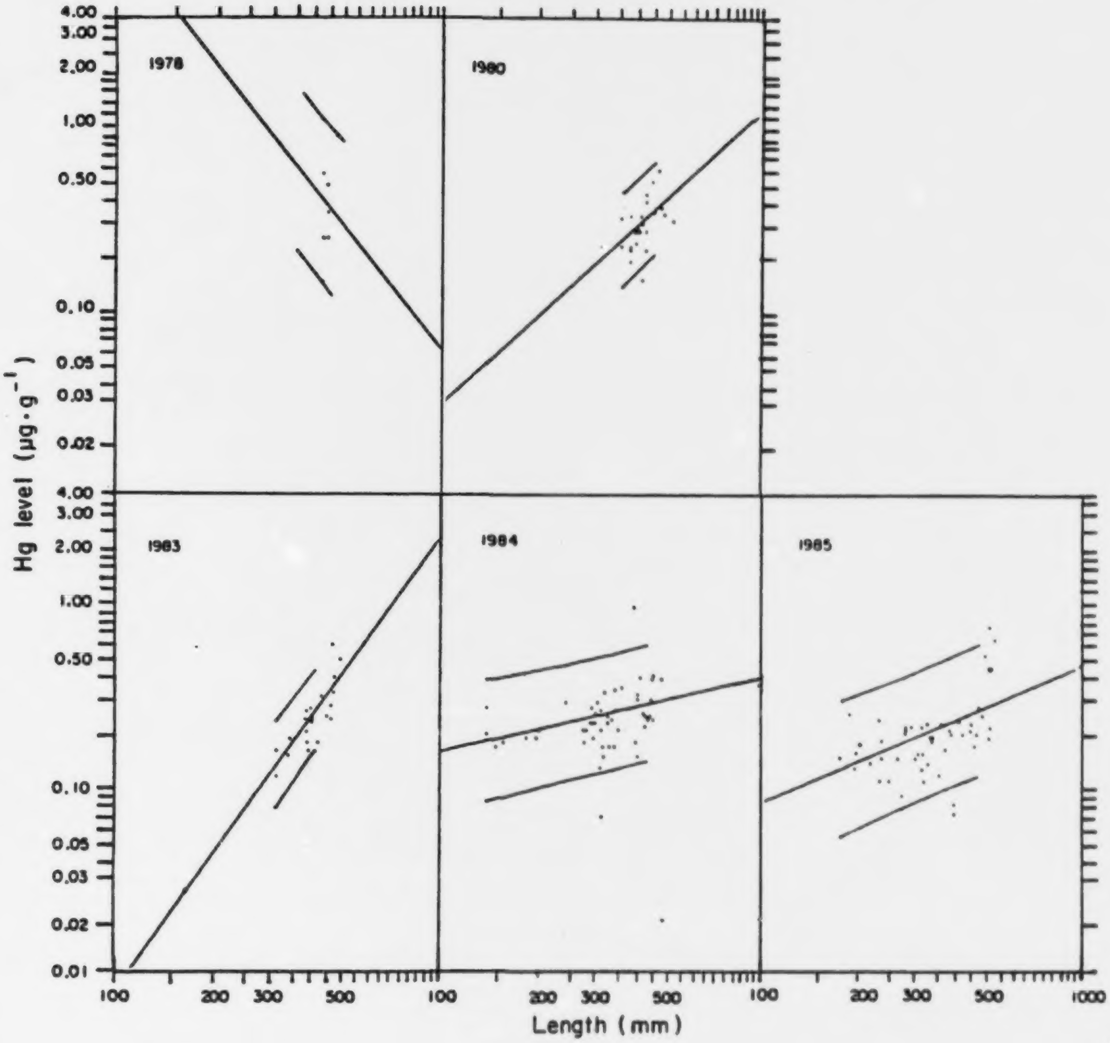

Fig. 12. Log - log regression lines and 95% confidence intervals for muscle mercury vs fork length relationships for whitefish samples taken from Rat Lake in 1978, 1980, and 1983 - 1985.

Although the regression line for the 1981 whitefish sample from Notigi Lake (Fig. 13.) diverged from those fitted to the 1983 and 1985 samples, the 1981 data were contained within the latters, 95% confidence intervals. Regression lines for the 1983 and 1985 Notigi Lake whitefish samples were similar in slope, but the 1985 line exhibited a slightly higher elevation. A similar circumstance was observed in the comparison of the 1983 and 1985 whitefish samples from Wapisu Lake (Fig. 14). There appeared to be considerable annual variation in the mercury levels of whitefish from Footprint Lake (Fig. 15). However, data points for the 1980, 1981 and 1983 samples generally were encompassed by the 95% confidence intervals about the 1985 sample regression. The 1984 Footprint Lake sample differed from the 1985 sample in that some whitefish in the 1984 which were less than 300 mm long exhibited lower mercury concentrations. Much the same observations were made for the sample regressions for Threepoint Lake whitefish, except that there was no appreciable difference between the 1984 and 1985 samples (Fig. 16). It is interesting to note that the trends in the regression line slopes for Footprint Lake and Threepoint Lake samples between 1983 and 1985 are the same, but the slopes tend to be steeper for whitefish from Threepoint Lake. This suggests the level of mercury contamination may be slightly higher in Threepoint Lake. Threepoint Lake is directly impacted by flows in the Diversion, while Footprint Lake is on a backflooded tributary.

The relationship of the earlier sample data for walleyes to that for walleyes collected between 1983 and 1985 was essentially the same as that described for whitefish, with the exception that walleyes exhibited much higher mercury levels. The 1978 and 1979 walleye samples from Rat Lake showed very high mercury concentrations ($\geq 2\ \mu g.g^{-1}$),

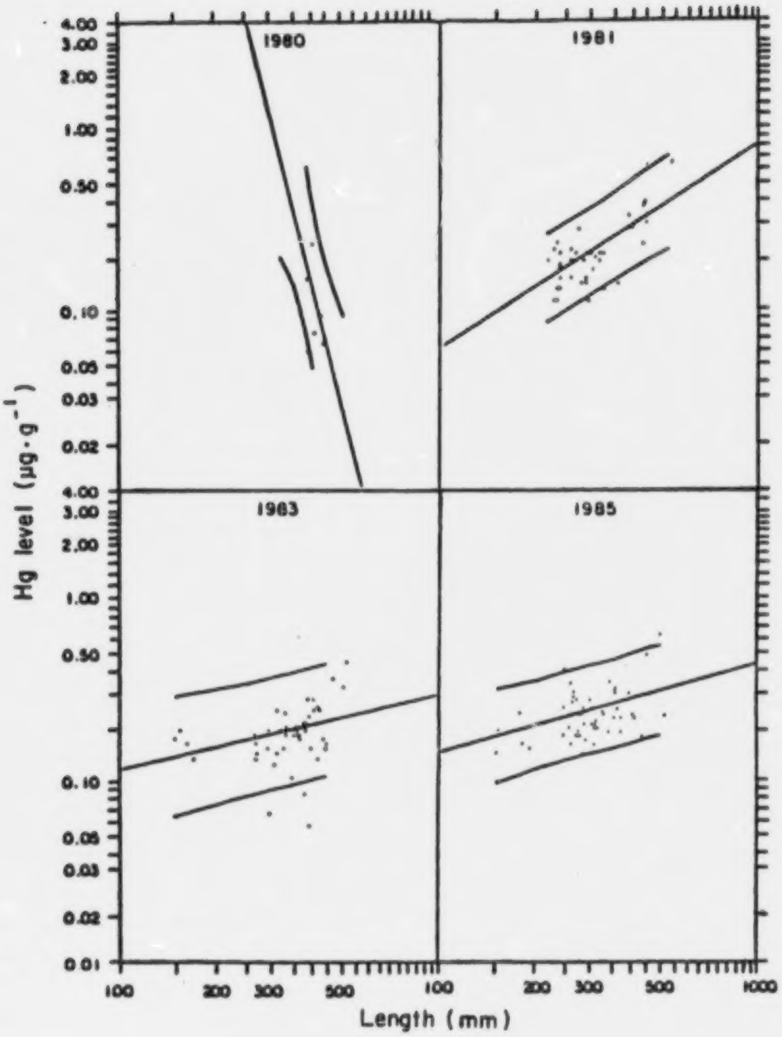

Fig. 13. Log - log regression lines and 95% confidence intervals for muscle mercury vs fork length relationships for whitefish samples taken from Notigi Lake in 1980, 1981, 1983 and 1985.

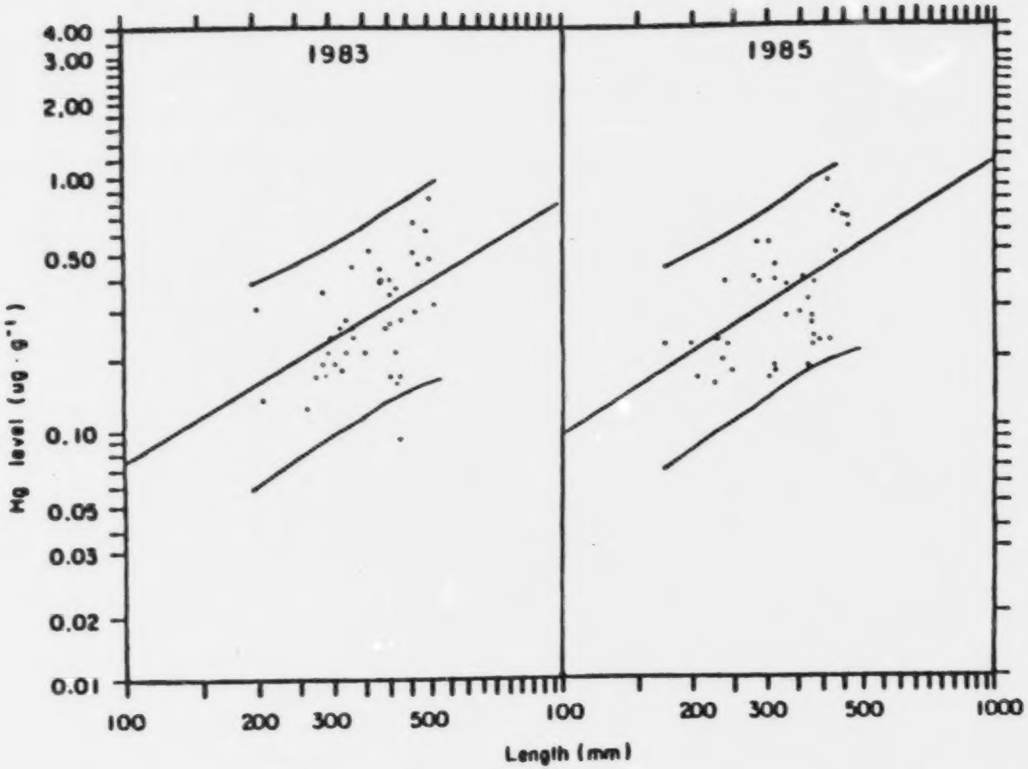

Fig. 14. Log - log regression lines and 95% confidence intervals for muscle mercury *vs* fork length relationships for whitefish samples taken from Wapisu Lake in 1983 and 1985.

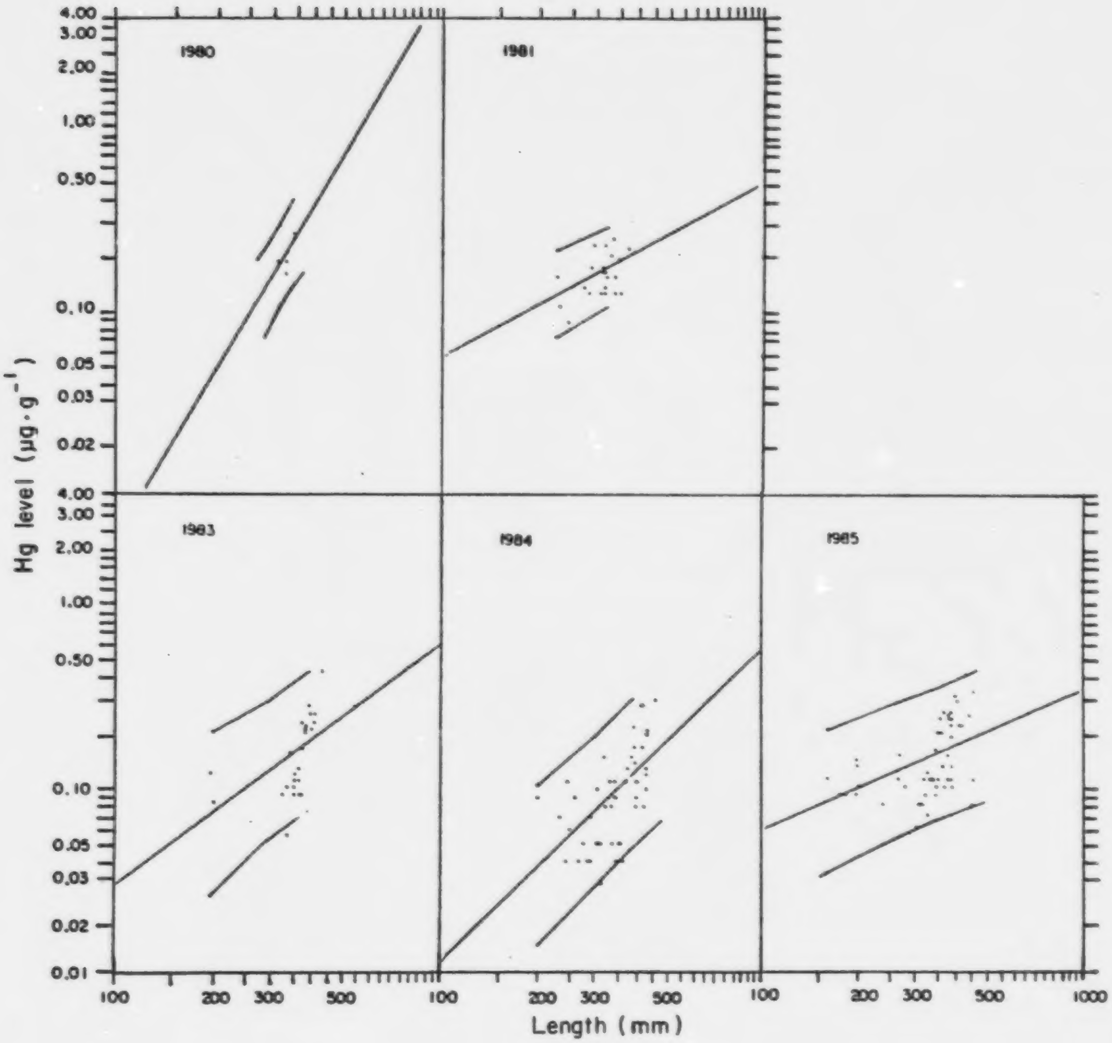

Fig. 15. Log - log regression lines and 95% confidence intervals for muscle mercury vs fork length relationships for whitefish samples taken from Footprint Lake in 1980, 1981 and 1983 - 1985.

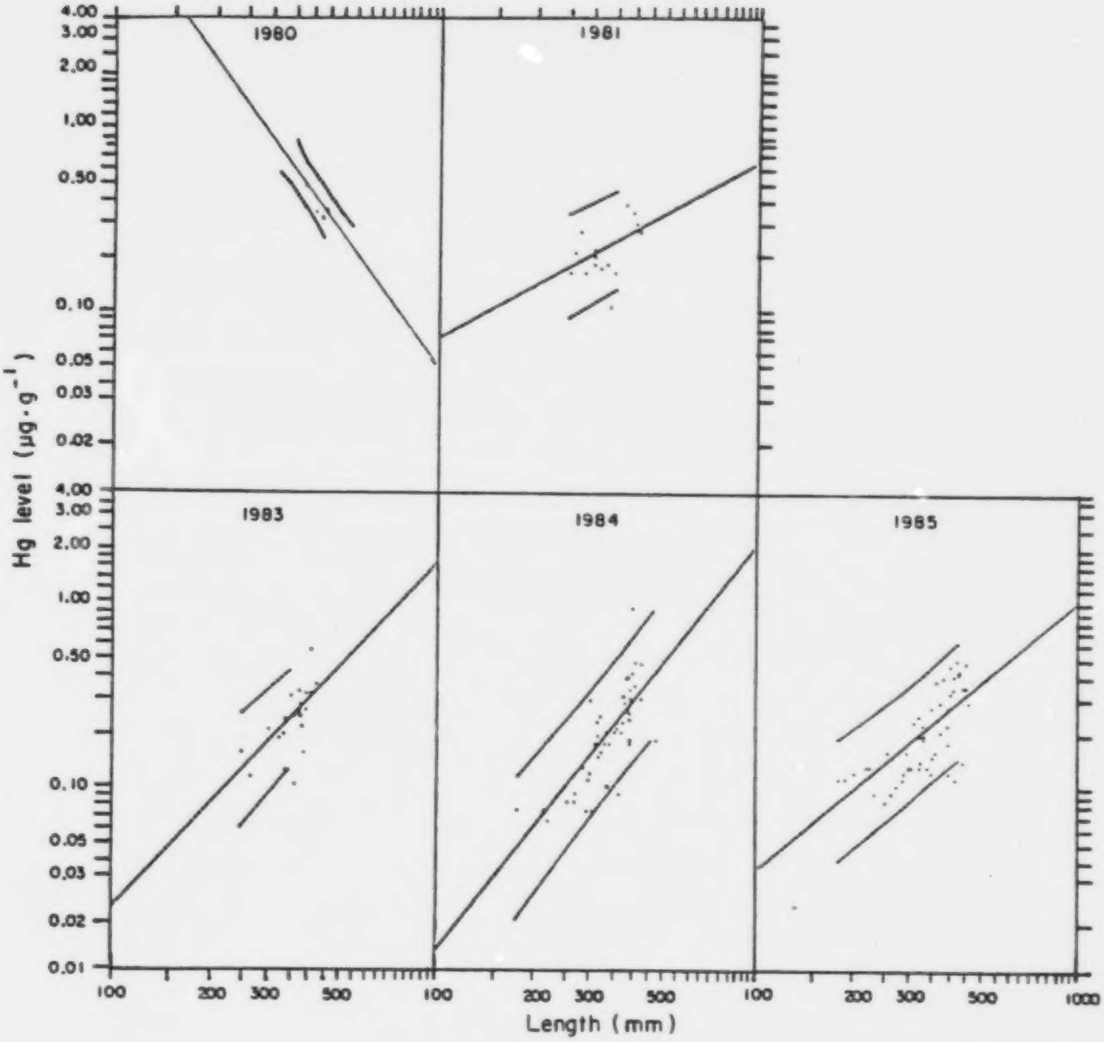

Fig. 16. Log - log regression lines and 95% confidence intervals for muscle mercury vs fork length relationships for whitefish samples taken from Threepoint Lake in 1980, 1981 and 1983 - 1985.

but these points fell well within the 95% confidence intervals for the 1984 and 1985 sample regressions (Fig. 17). The 1980 Rat Lake walleye sample differed from the 1983 - 1985 samples since the 1980 data tended to lie outside the lower band of the confidence limits for the 1983 - 1985 samples. For the Notigi Lake walleye (Fig. 18), most of the data points for the 1980, 1981 and 1982 samples lie within the regression line confidence intervals for both the 1983 and 1985 samples. Of special interest is the divergence in mercury levels in the 1978 sample. The upper cluster of points for this sample falls within the confidence belts for the 1983 sample regression. The lower cluster, however, does not have any points in common with any other walleye samples from Notigi Lake.

The 1977 walleye sample from Wapisu Lake was represented by individuals from a restricted size range; most were from 350 to 450 mm in length. The mercury levels in these fish ranged from less than 0.10 $\mu g \cdot g^{-1}$ to approximately 3 $\mu g \cdot g^{-1}$ (Fig. 19). Many of these data points fell within the 95% confidence intervals from the 1983 regression. Those 1977 data points which were outside the 1983 confidence belts exhibited low mercury levels. The 1983 and 1985 sample regressions for Wapisu Lake walleye were similar, except that the 1983 sample showed a much greater variability in mercury levels.

The limited number of data points for the 1978, 1980 and 1981 walleye samples from Footprint Lake, all of which were represented by fish of a limited size range, fell within the 95% confidence interval belts for the 1983 - 1985 sample regressions (Fig. 20). The 1983 sample regression was almost identical to that for the 1985 sample, but as in

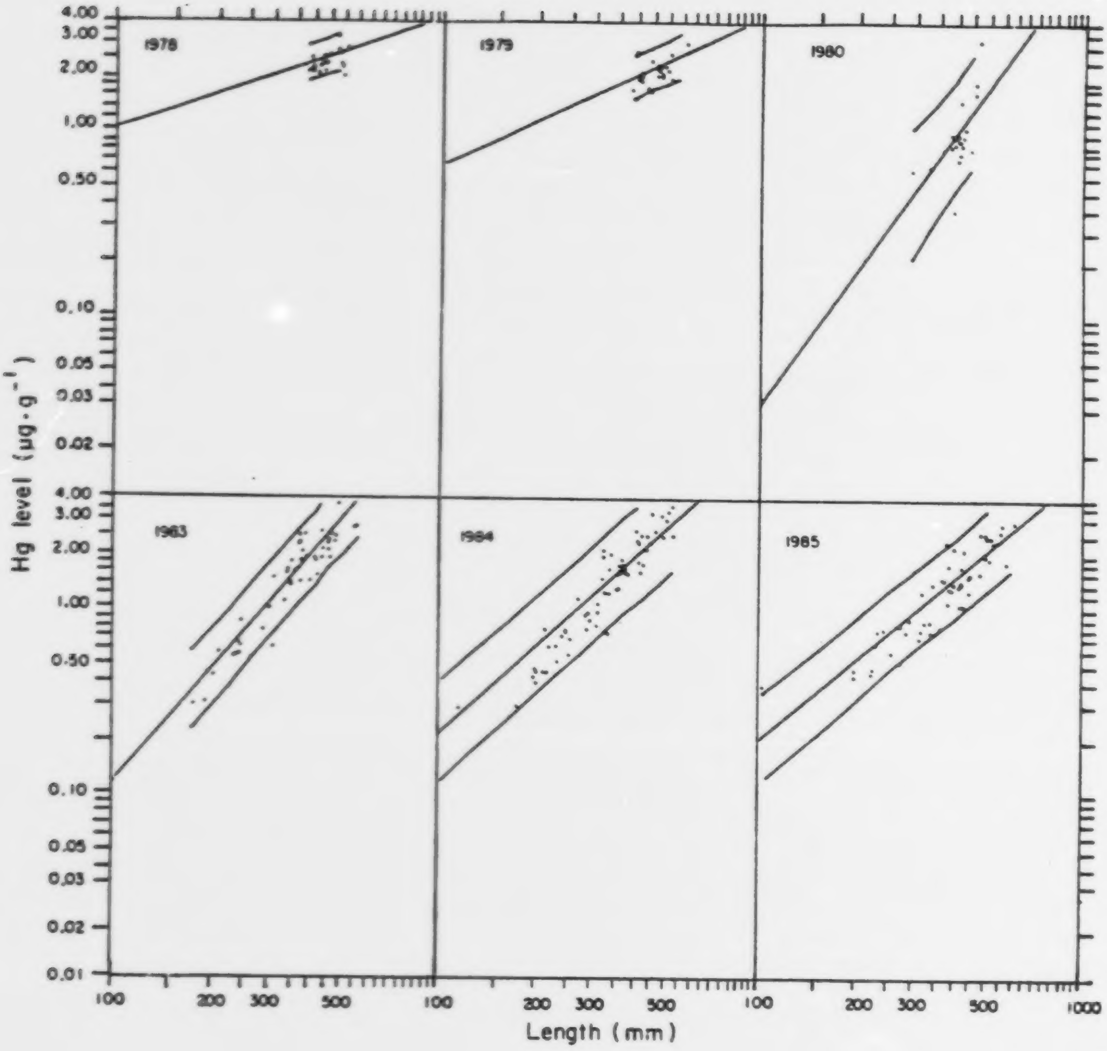

Fig. 17. Log - log regression and 95% confidence intervals for muscle mercury vs fork length relationships for walleye samples taken from Rat Lake in 1978 - 1980 and 1983 - 1985.

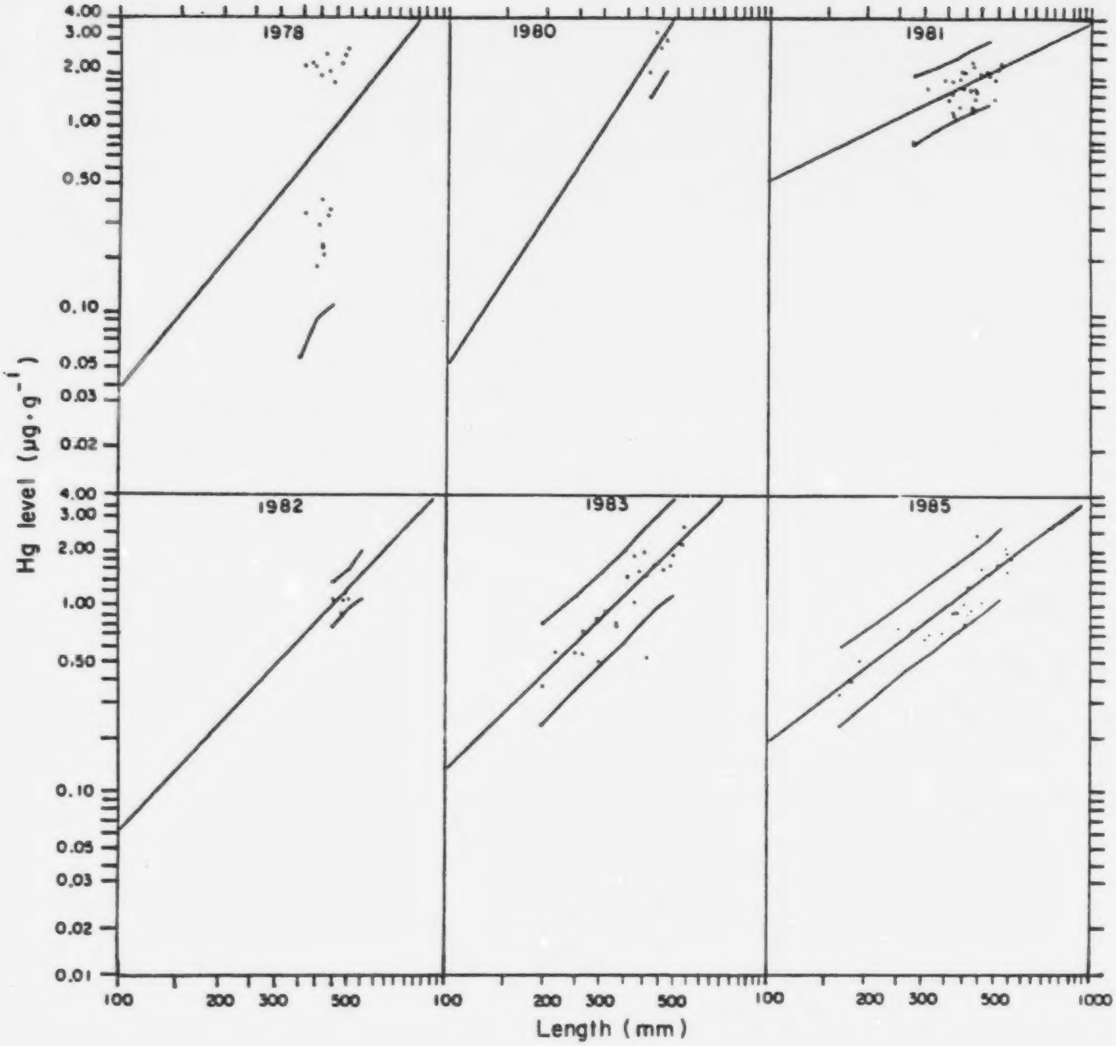

Fig. 18. Log - log regression lines and 95% confidence intervals for muscle mercury vs fork length relationships for walleye samples taken from Notigi Lake in 1978, 1980 - 1983 and 1985.

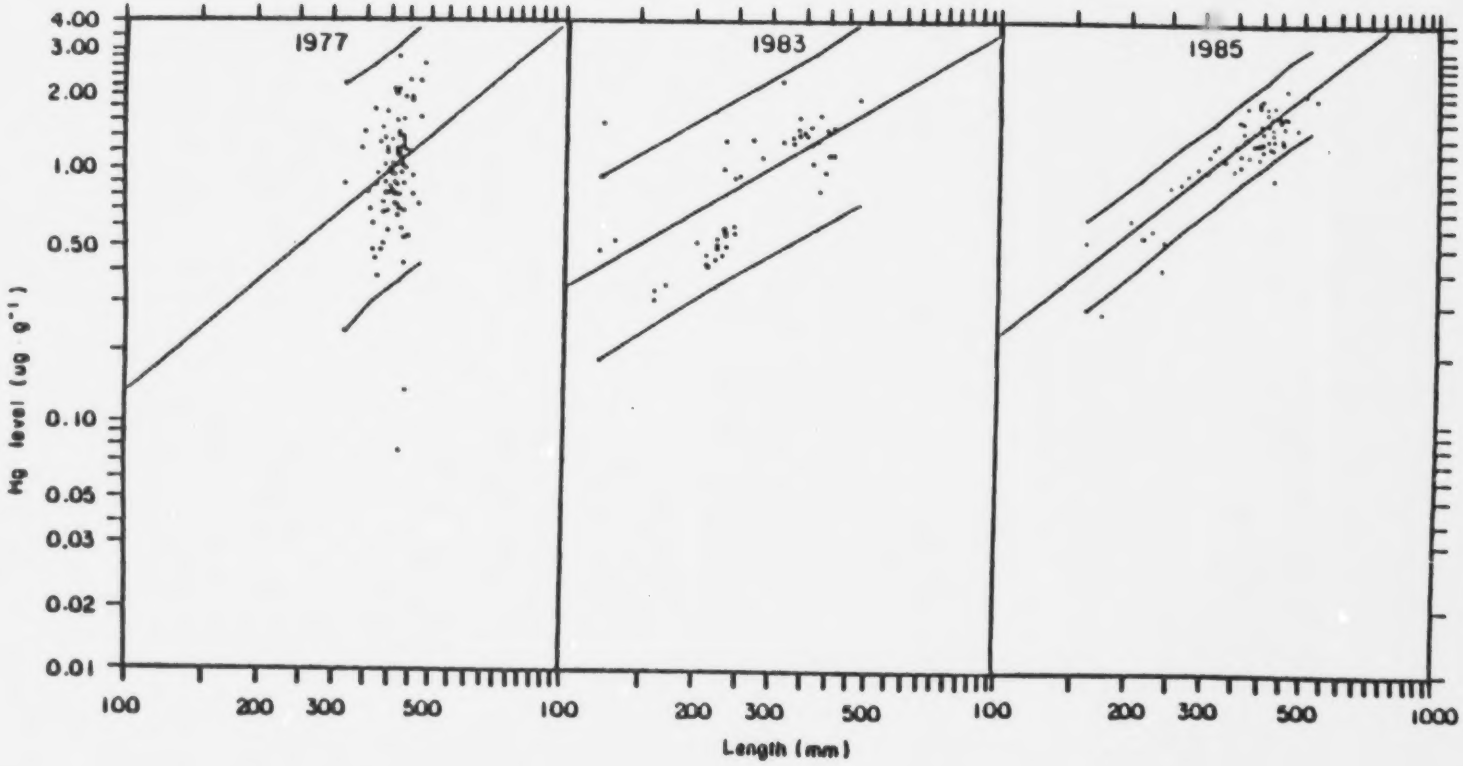

Fig. 19. Log - log regression lines and 95% confidence intervals for muscle mercury vs fork length relationships for walleye samples taken from Wapisu Lake in 1977, 1983 and 1985.

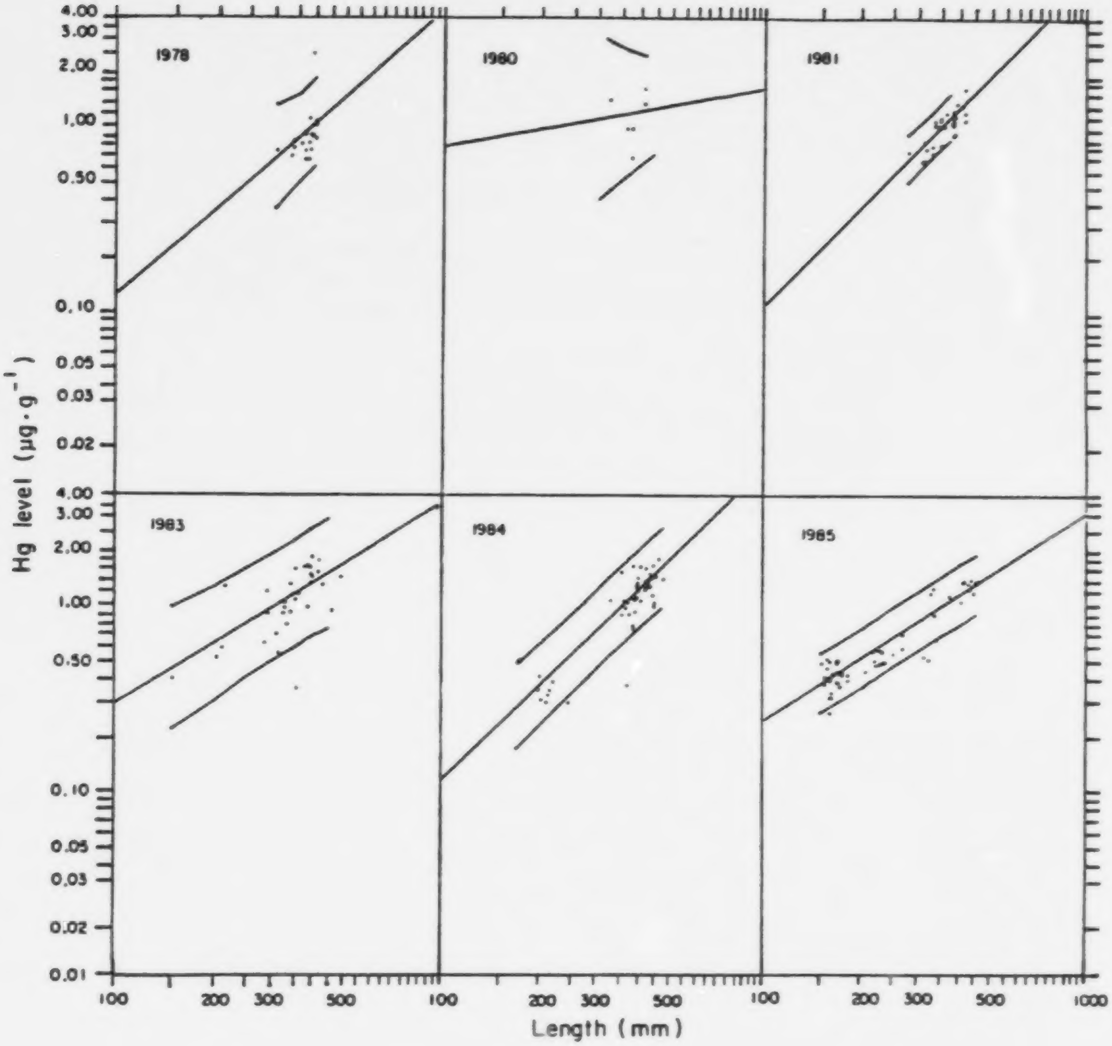

Fig. 20. Log - log regression lines and 95% confidence intervals for muscle mercury vs fork length relationships for walleye samples taken from Footprint Lake in 1978, 1980, 1981 and 1983 - 1985.

the case of the walleye from Wapisu Lake there was greater variability in the mercury concentrations of the 1983 sample. In walleye from Threepoint Lake, data points for the 1980 and 1981 samples were also within the confidence intervals for the 1983 - 1985 sample regressions (Fig. 21). The regression line slope for the 1983 sample was identical to that of the 1984 and 1985 samples, but its elevation was higher. The regression lines and confidence intervals for the 1984 and 1985 walleye samples from Threepoint Lake were virtually the same. The 1983 - 1985 sample regressions for Threepoint Lake also approximated those determined for nearby Footprint Lake walleye .

The longest sequence of walleye mercury data (∿ 12 years) was available for Sipiwesk lakes and Split lakes. The distribution of data points for walleye mercury samples from Sipiwesk Lake were similar in most years (Fig. 22a) and with some exceptions, the data points generally fell within the 95% confidence limits for the 1983 sample regression shown in Fig. 22b . The exceptions were cases in which large samples of walleye over a limited size range were taken. In these samples (e.g. 1977 and 1981), a much greater variability in mercury levels in fish of a similar size was evident. Annual comparisons of the sample regressions indicated no evidence to suggest any positive or negative changes in the level of mercury contamination of Sipiwesk Lake walleye throughout the period of record. For example, the 1972 and 1976 sample data were very similar, and the 1976 data, in turn, approximated the 1983 data. Although there was considerable variation in the year to year configurations of regression lines and confidence intervals fitted to the Split Lake walleye mercury data, the relative distributions of data

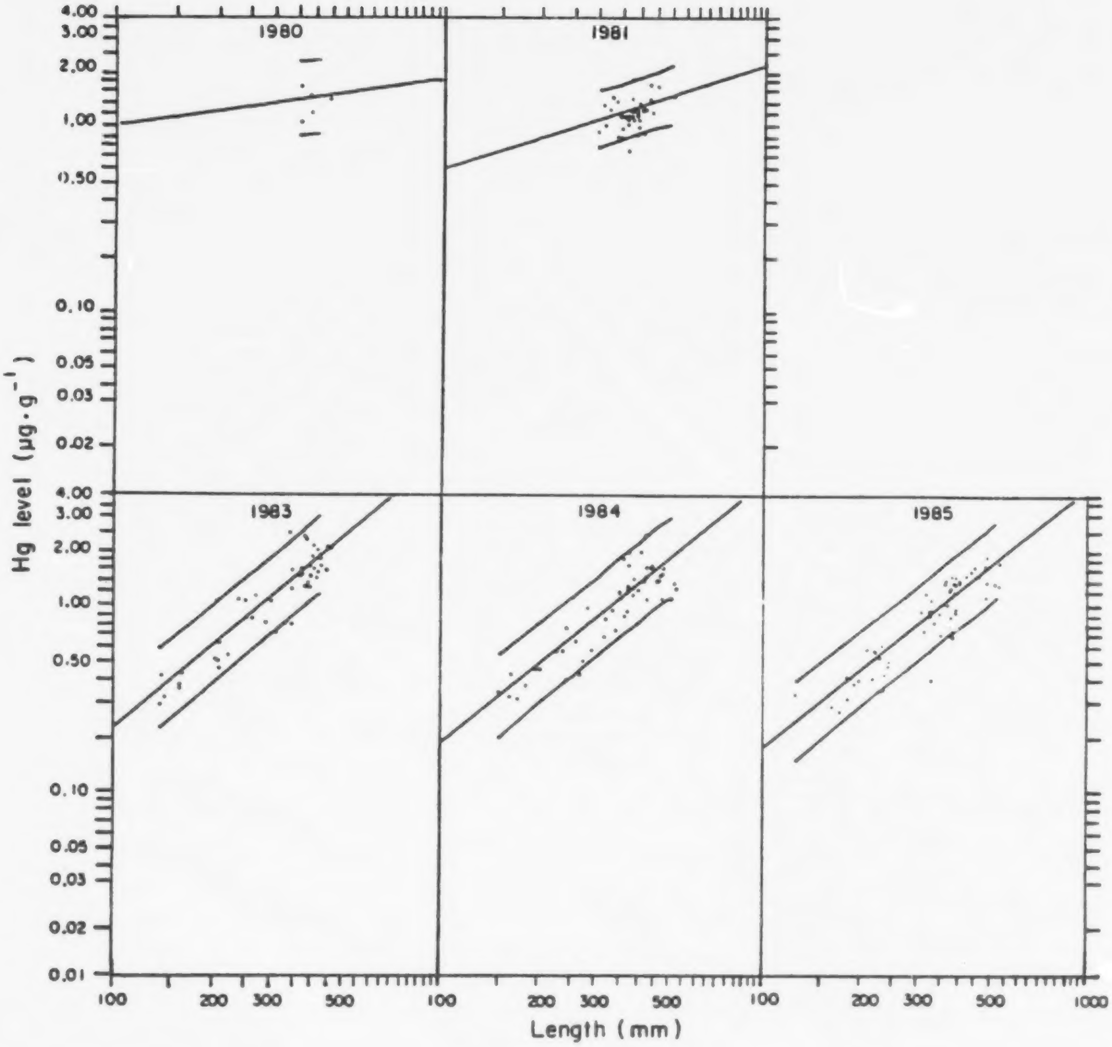

Fig. 21. Log - log regression lines and 95% confidence intervals for muscle mercury vs fork length relationships for walleye samples taken from Threepoint Lake in 1980, 1981 and 1983 - 1985.

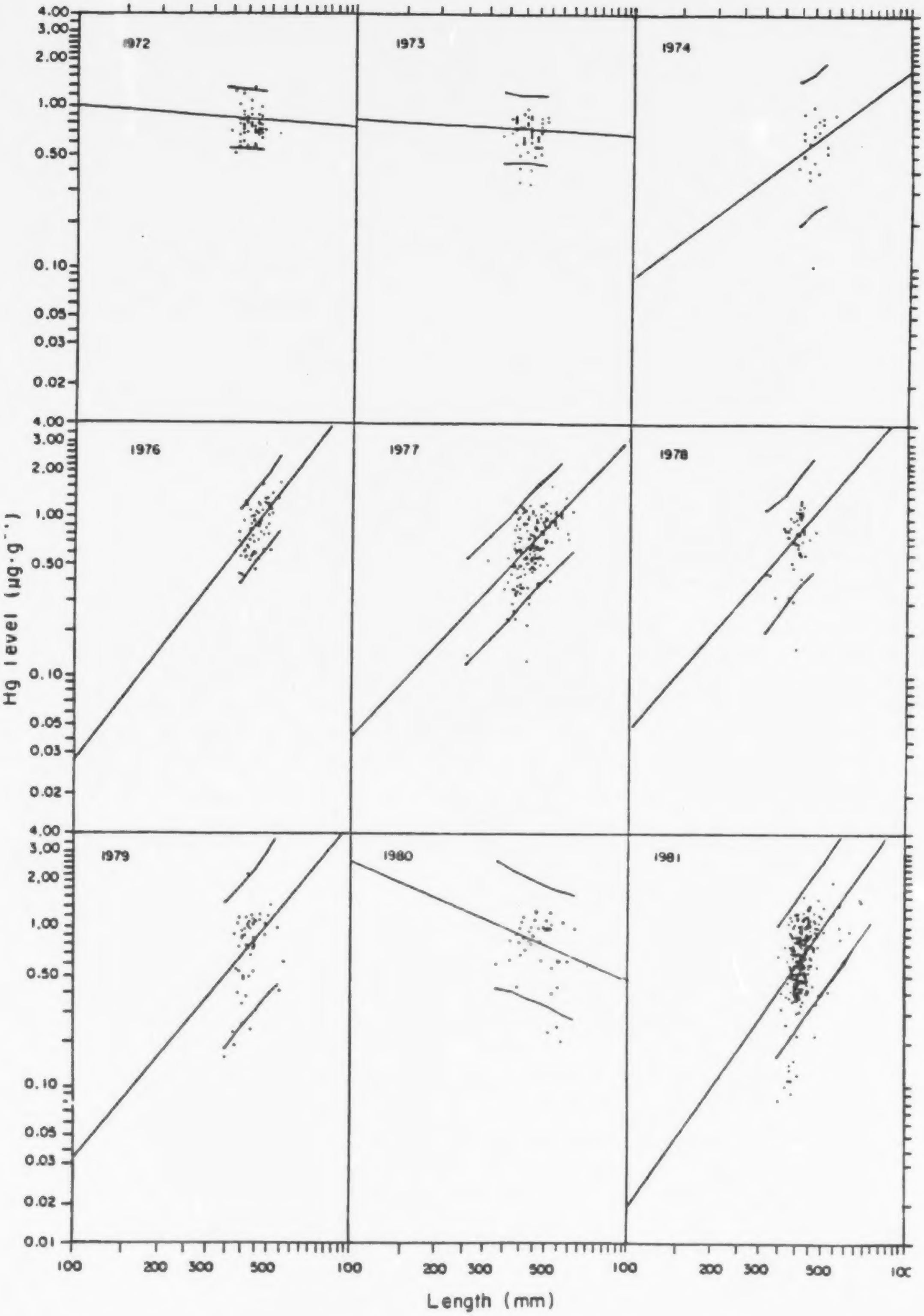

Fig. 22a. Log - log regression lines and 95% confidence intervals for muscle mercury vs fork length relationships for walleye samples taken from Sipiwesk Lake in 1972 - 1974 and 1976 - 1981.

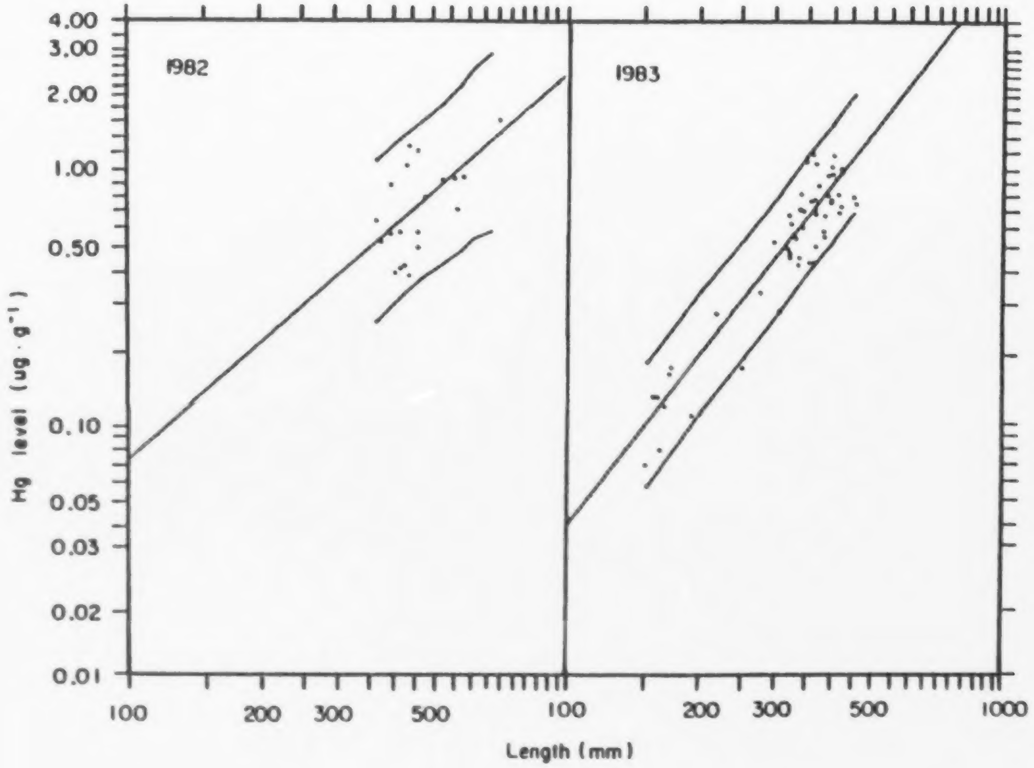

Fig. 22b. Log - log regression lines and 95% confidence intervals for muscle mercury vs fork length relationships for walleye samples taken from Sipiwesk Lake in 1982 and 1983.

points in all years were also basically the same (Fig. 23a and b). Walleye in the 1983 sample from Split Lake (Fig. 23b) were from a limited size range, but again most of the data points were within the 95% confidence interval of the 1985 sample regression.

Mercury concentrations in northern pike sampled from Notigi forebay (i.e. Rat and Notigi lakes) prior to 1983 were similar to those observed in pike sampled from 1983 to 1985; that is, data points for the pre -1983 samples generally fell within the 95% confidence intervals for regression lines fitted to the most recent sample data (Figs. 24 and 25). Regression lines for the 1984 and 1985 pike sample data from Rat Lake were almost indentical (Fig. 24). The 1985 sample of northern pike from Notigi Lake, although consisting of individuals of a limited size range, was similar to the 1982 sample. Data points for the 1985 samples were contained within the 95% confidence zone about the 1983 sample regression line (Fig. 25). The regression line for the 1983 pike sample from Notigi Lake was similar to the 1985 sample regression for pike from Rat Lake.

Among the northern pike samples from Wapisu Lake, less than half of the data points for the 1977 sample were within the confidence belts of either the 1983 or 1985 sample regressions (Fig. 26). Most of the individual pike in the 1977 sample had lower mercury levels than pike of equivalent sizes in the 1983 or 1984 samples. Although the 1983 sample of Wapisu Lake pike appeared to show a higher level of mercury contamination than the 1985 sample, this apparently may again be due to the limited size range of fish in the 1983 sample. Inspection of the data points indicated that the majority of points for the 1983 sample fell within the relatively narrow confidence intervals about the 1985

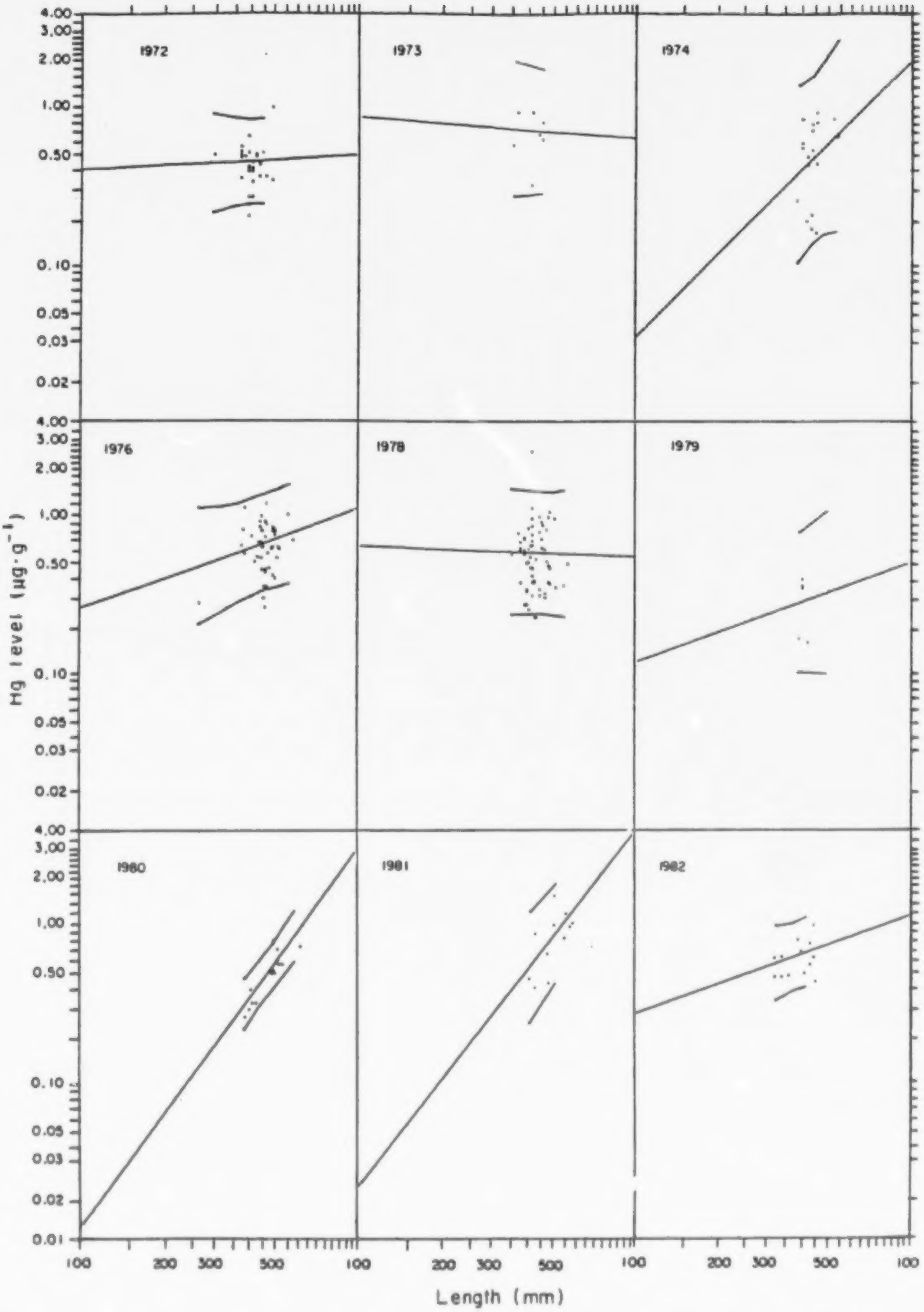

Fig. 23a. Log - log regression lines and 95% confidence intervals for muscle mercury vs fork length relationships for walleye samples taken from Split Lake in 1972 - 1974, 1976 and 1978 - 1982.

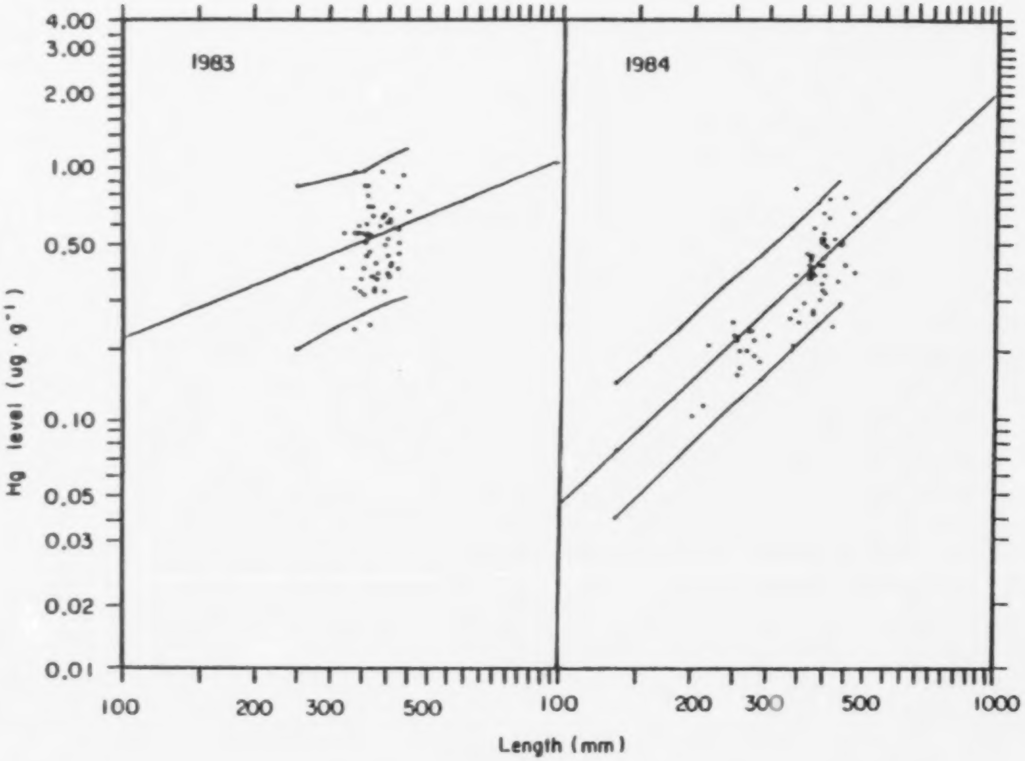

Fig. 23b. Log - log regression lines and 95% confidence intervals for muscle mercury vs fork length relationships for walleye samples taken from Split Lake in 1983 and 1984.

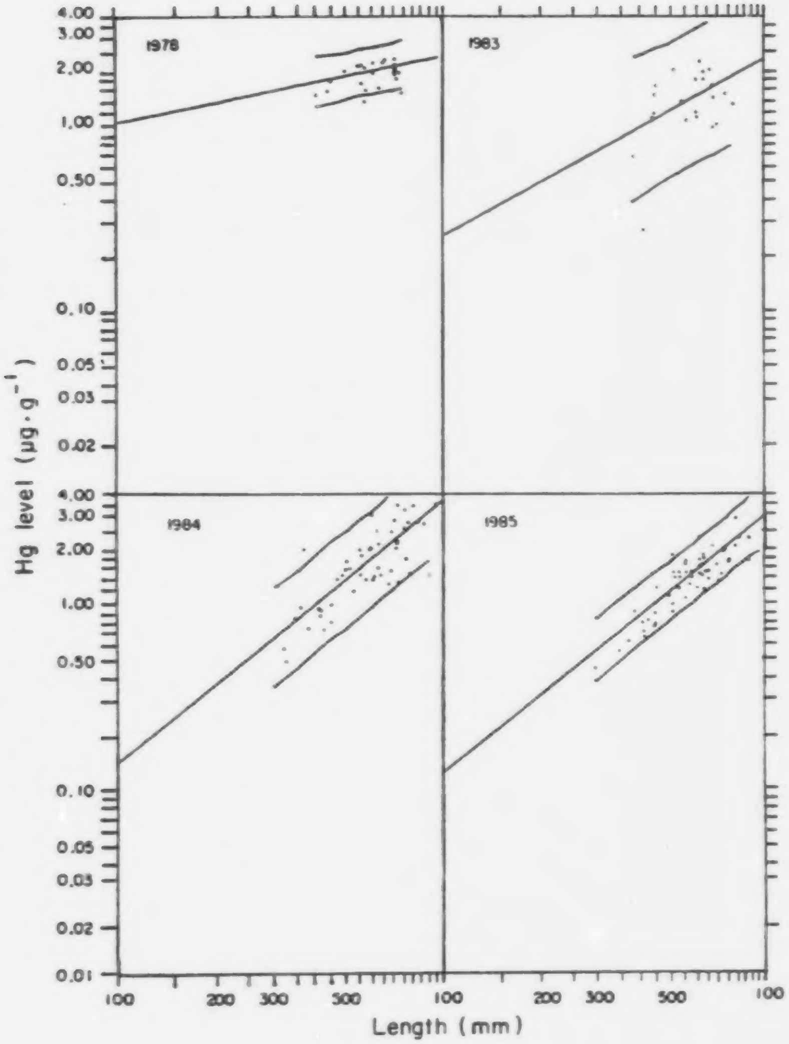

Fig. 24. Log - log regression lines and 95% confidence intervals for muscle mercury vs fork length relationships for northern pike samples taken from Rat Lake in 1978 and 1983 - 1985.

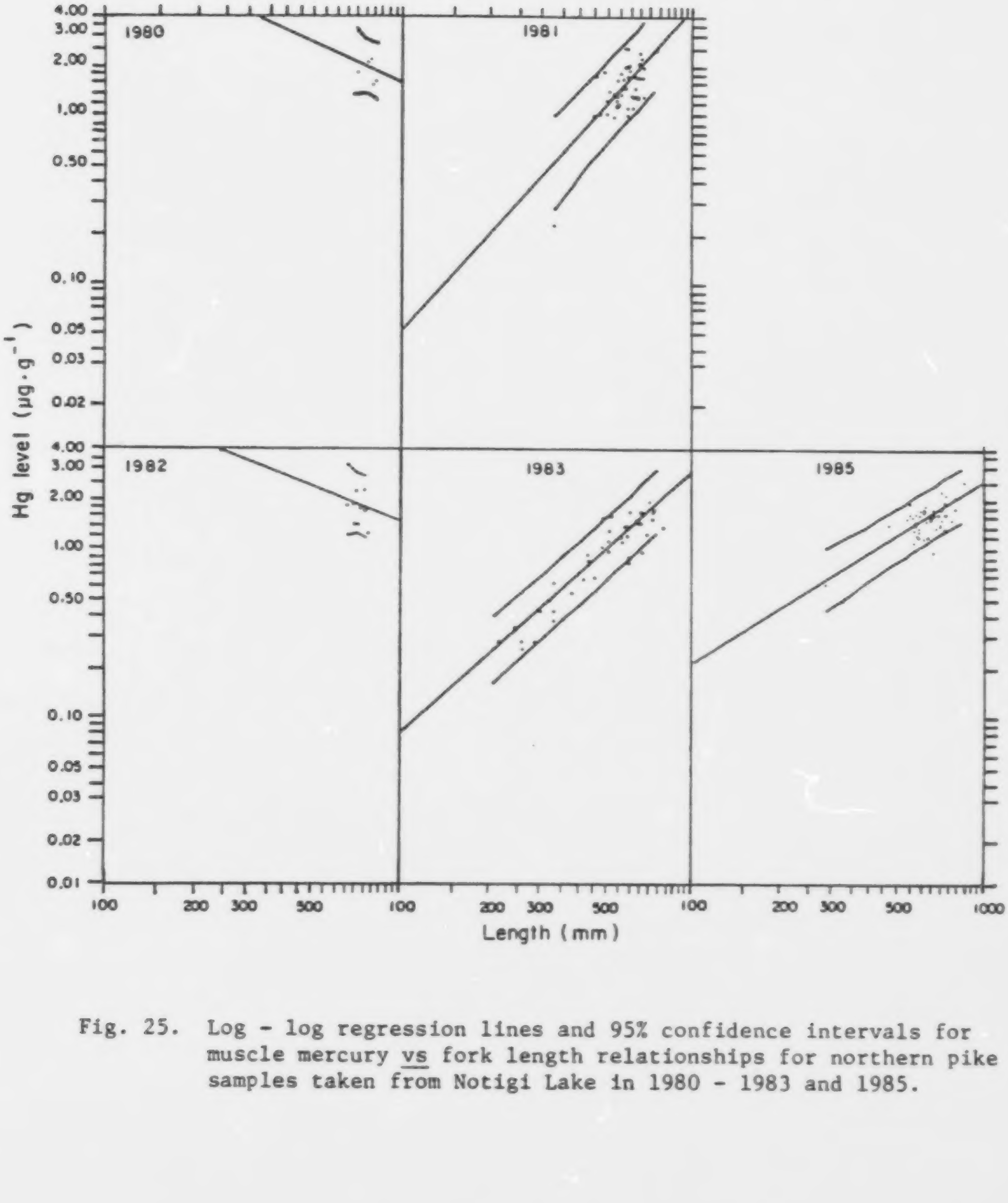

Fig. 25. Log - log regression lines and 95% confidence intervals for muscle mercury vs fork length relationships for northern pike samples taken from Notigi Lake in 1980 - 1983 and 1985.

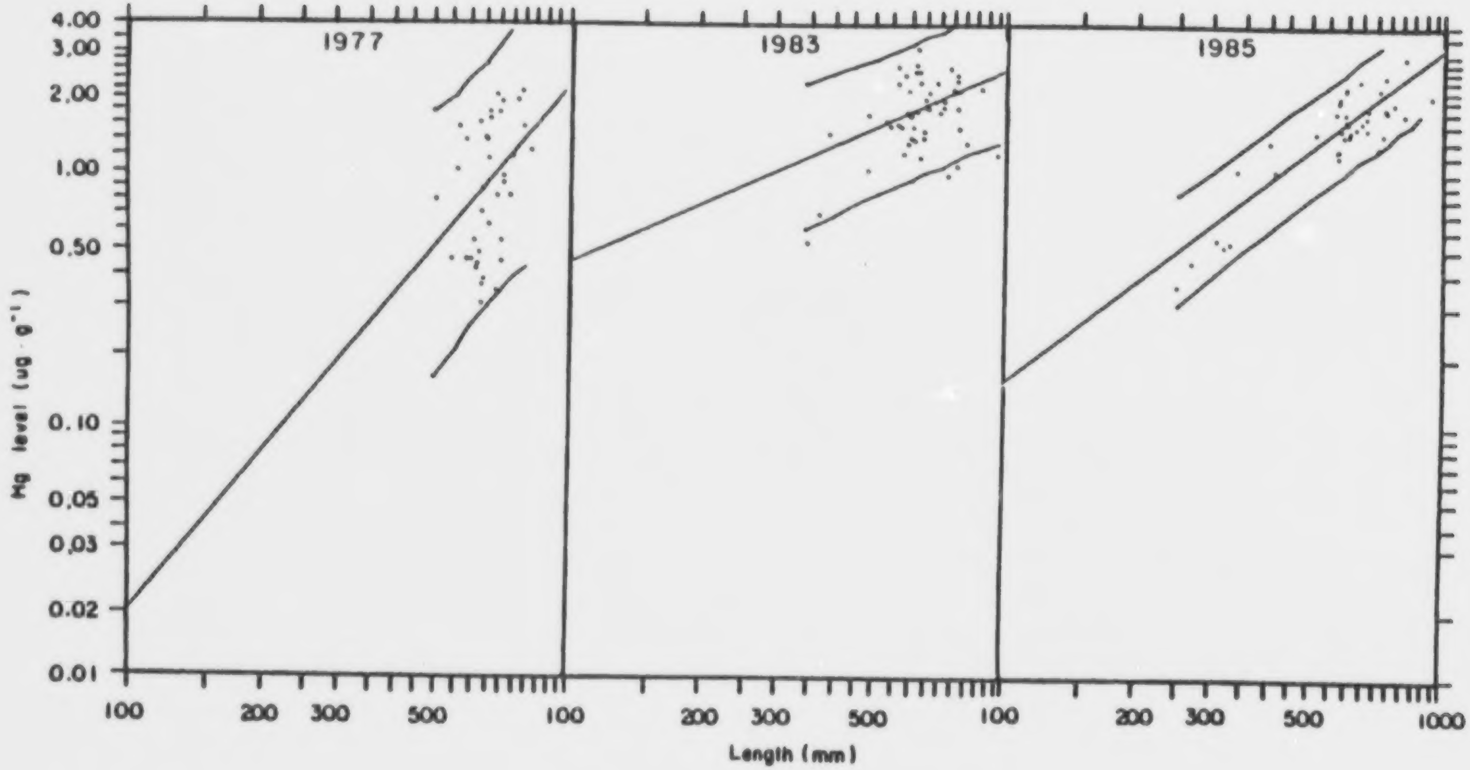

Fig. 26. Log - log regression lines and 95% confidence intervals for muscle mercury vs fork length relationships for northern pike samples taken from Wapisu Lake in 1977, 1983 and 1985.

sample regression line. Comparison of the pike mercury data for Wapisu Lake to data for upstream lakes in the Notigi forebay and downstream lakes in the Nelson House area revealed a closer similarity to pike mercury levels in the Notigi forebay.

Observations similar to those made above for northern pike samples from the Notigi forebay lakes were also made for pike mercury samples from Footprint and Threepoint lakes in the Nelson House area. With a few exceptions, data points for samples taken before 1983 were inscribed by the confidence intervals for the 1983 - 1985 regressions (Fig. 27 and 28). There was also some similarity in the 1983 - 1985 sample regressions between Footprint Lake and Threepoint Lake.

Graphic comparisons of mercury data for northern pike sampled from Sipiwesk Lake and Split Lake were much like those discussed for walleye samples from the same lakes. The distributions of data points for annual pike samples from Sipiwesk Lake (Fig. 29a) were similar and in most cases were included in the confidence intervals for the 1983 sample regression (Fig. 29b). It is interesting to note that despite the restricted size ranges of fish in most samples, sample regression lines were almost identical in the following comparisons: 1972 - 1980, 1972 - 1981, 1976 - 1979, 1980 - 1981 and 1981 - 1982. The 1982 Sipiwesk Lake pike sample exhibited a large variation in mercury concentration (approximately 0.14 to 2.19 $\mu g.g^{-1}$) for individuals of a similar size, but most of the data points nevertheless were contained within the 95% confidence intervals for the 1983 sample regression which was based on fish of a wider size range than represented in the 1982 sample. With reference to Split Lake northern pike, the distribution of data points

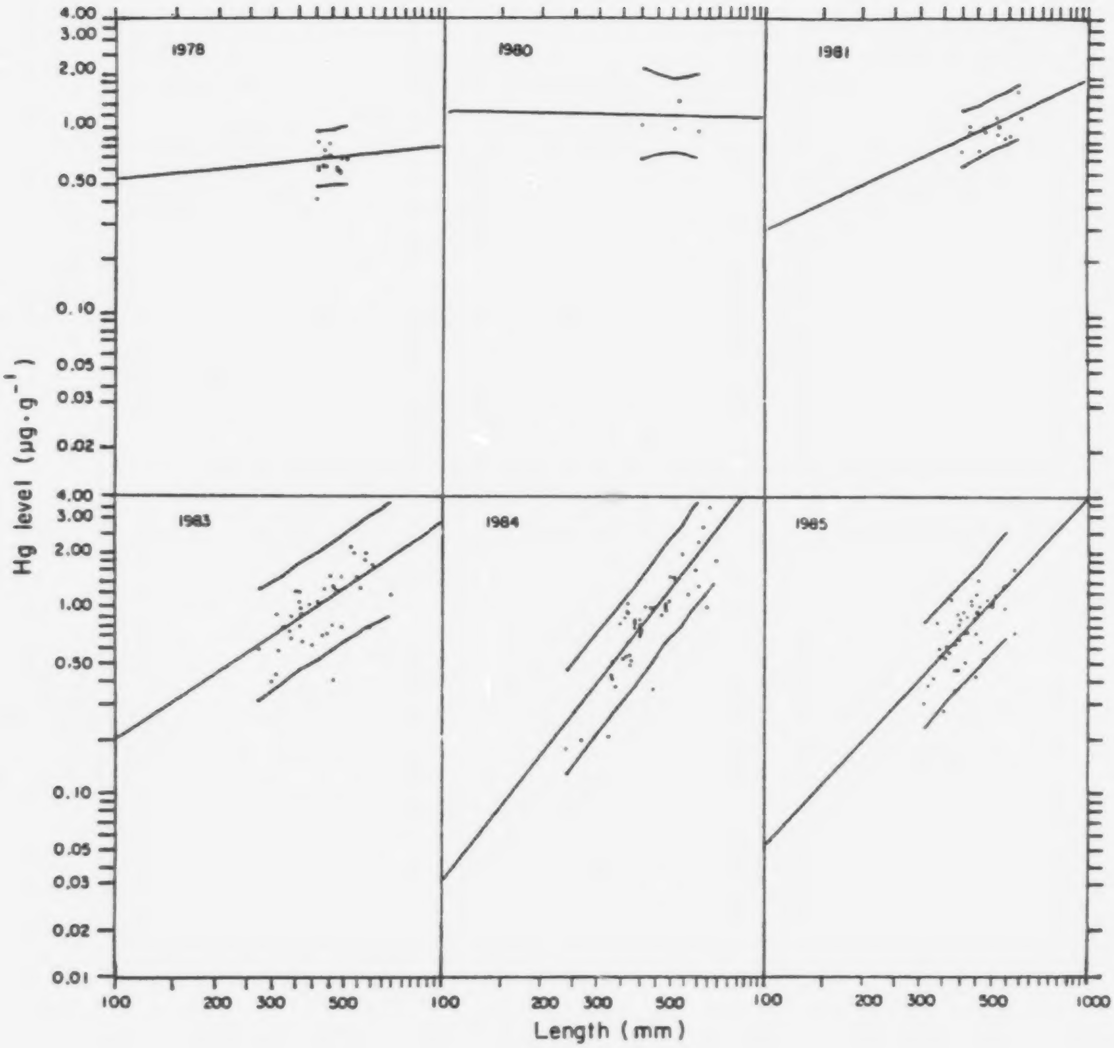

Fig. 27. Log - log regression lines and 95% confidence intervals for muscle mercury vs fork length relationships for northern pike samples taken from Footprint Lake in 1978, 1980, 1981 and 1983 - 1985.

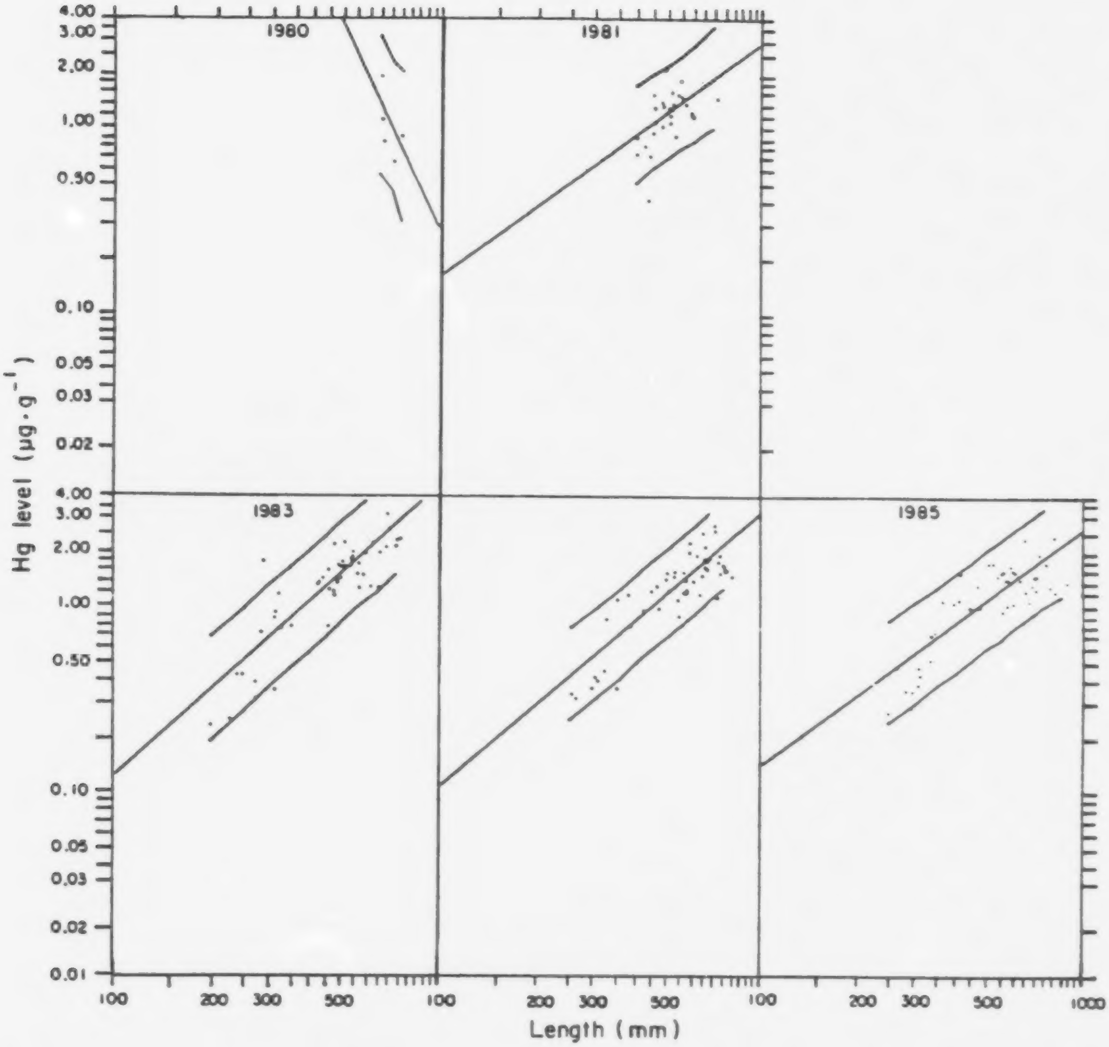

Fig. 28. Log - log regression lines and 95% confidence intervals for muscle mercury vs fork length relationships for northern pike samples taken from Threepoint Lake in 1980, 1981 and 1983 - 1985.

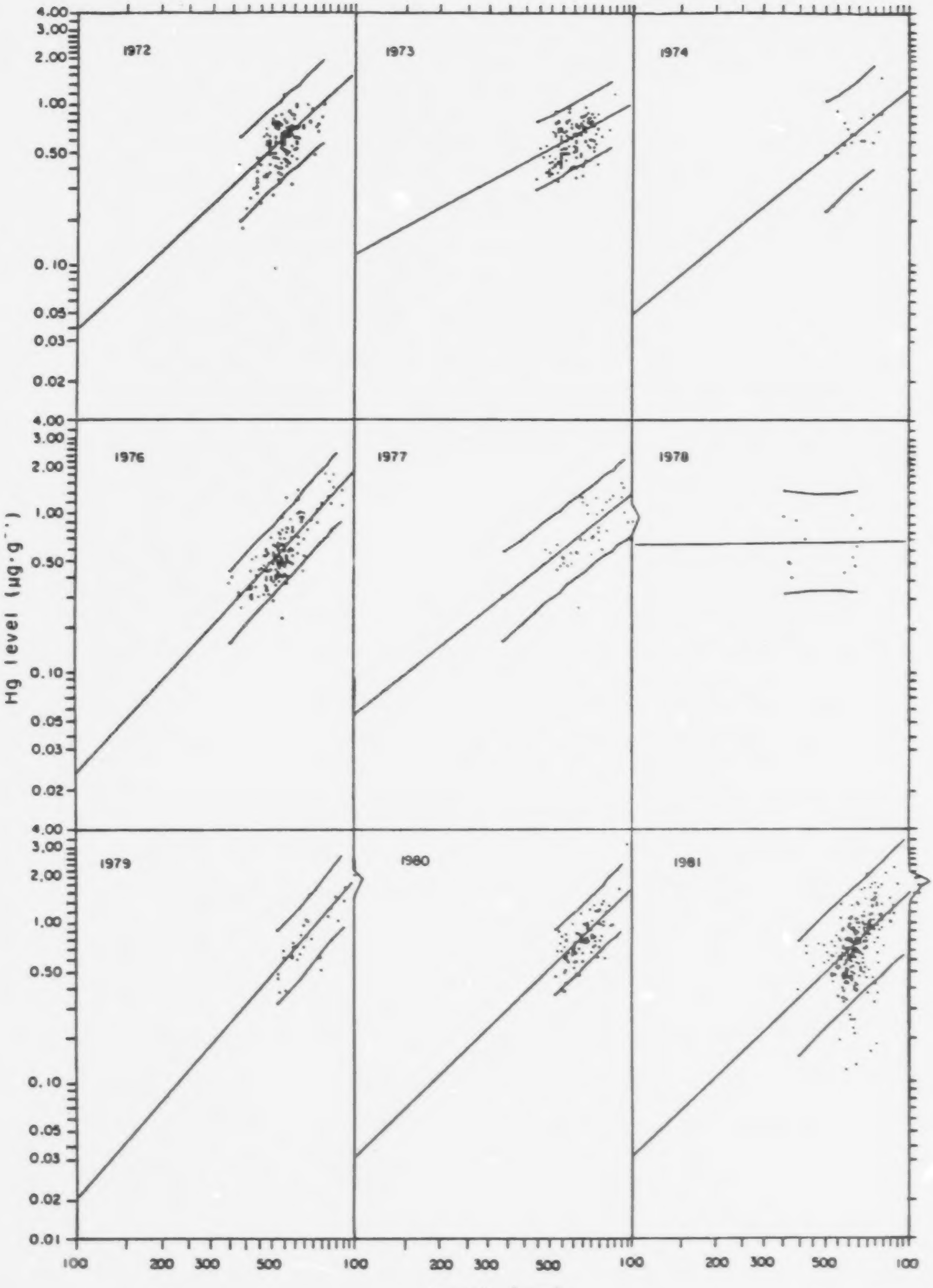

Fig. 29a. Log - log regression lines and 95% confidence intervals for muscle mercury vs fork length relationships for northern pike samples taken from Sipiwesk Lake in 1972 - 1974 and 1976 - 1981.

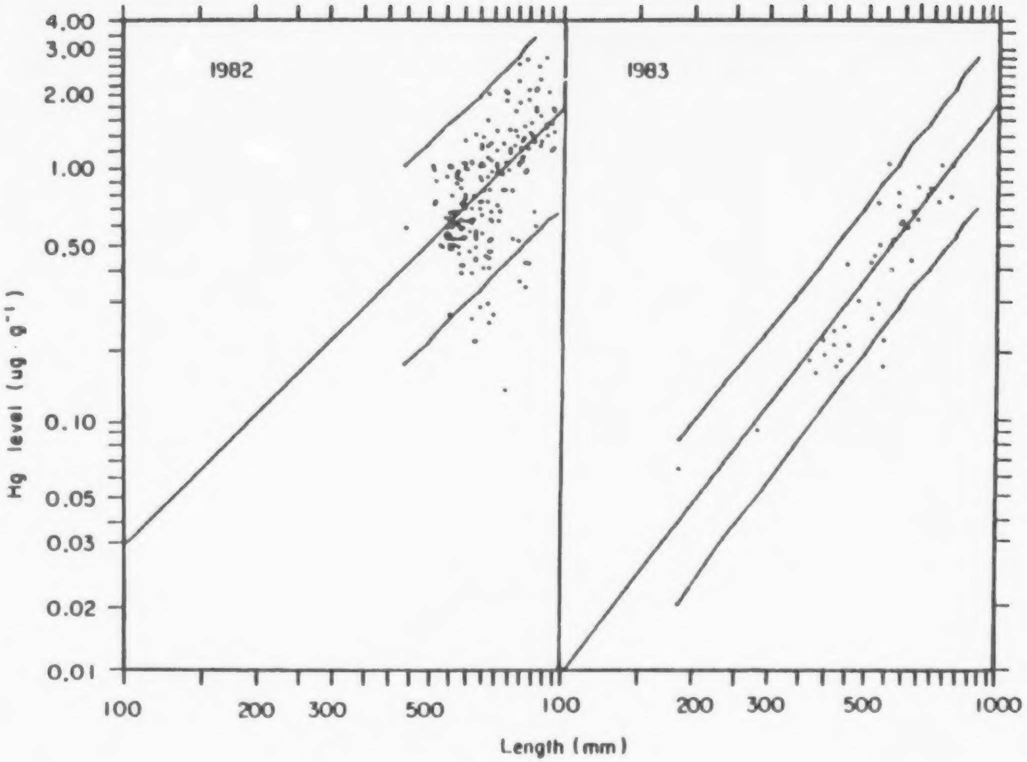

Fig. 29b. Log - log regression lines and 95% confidence intervals for muscle mercury vs fork length relationships for northern pike samples taken from Sipiwesk Lake in 1982 and 1983.

for the 1972 - 1982 samples were much the same relative to the confidence intervals for the 1983 and 1984 sample regression lines (Fig. 30). The conclusion that there have been no significant temporal changes in the level of mercury contamination in northern pike in Sipiwesk and Split lakes is supported by the observations that there are no marked trends in the average mercury concentrations in either survey or commercial samples[1] of pike from these lakes over a 16 year period (Fig. 31).

Comparisons of the log - log regression lines on a lake to lake basis for whitefish, walleye and northern pike samples taken from 1983 to 1985, showed striking similarities. For example, the configuration of regression lines for whitefish sample taken in 1983 (Fig. 32) closely resembled the pattern for the 1983 walleye samples (Fig. 33) and the 1983 northern pike samples (Fig. 34). The results for 1985 are especially interesting in that they compare the level of mercury contamination in lakes on the Churchill River Diversion to two non-flooded lakes (Costello and Suwannee lakes) in the same drainage basin and one unimpacted lake (Assean Lake) which is tributary to Split Lake. Regression lines for the 1985 walleye (Fig. 33) and pike samples (Fig. 34) from lakes on the Churchill River Diversion are tightly grouped and are distinctly separated from those for the unimpacted lakes. The same relationship is also evident for whitefish (Fig. 32), except that the regression lines are not as closely grouped.

Comparisons of length adjusted mean mercury levels in

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[1] In survey samples, mercury analyses are conducted on individual fish. In commercial samples, mercury analysis is done on the homogenate of muscle tissue from several or more fish.

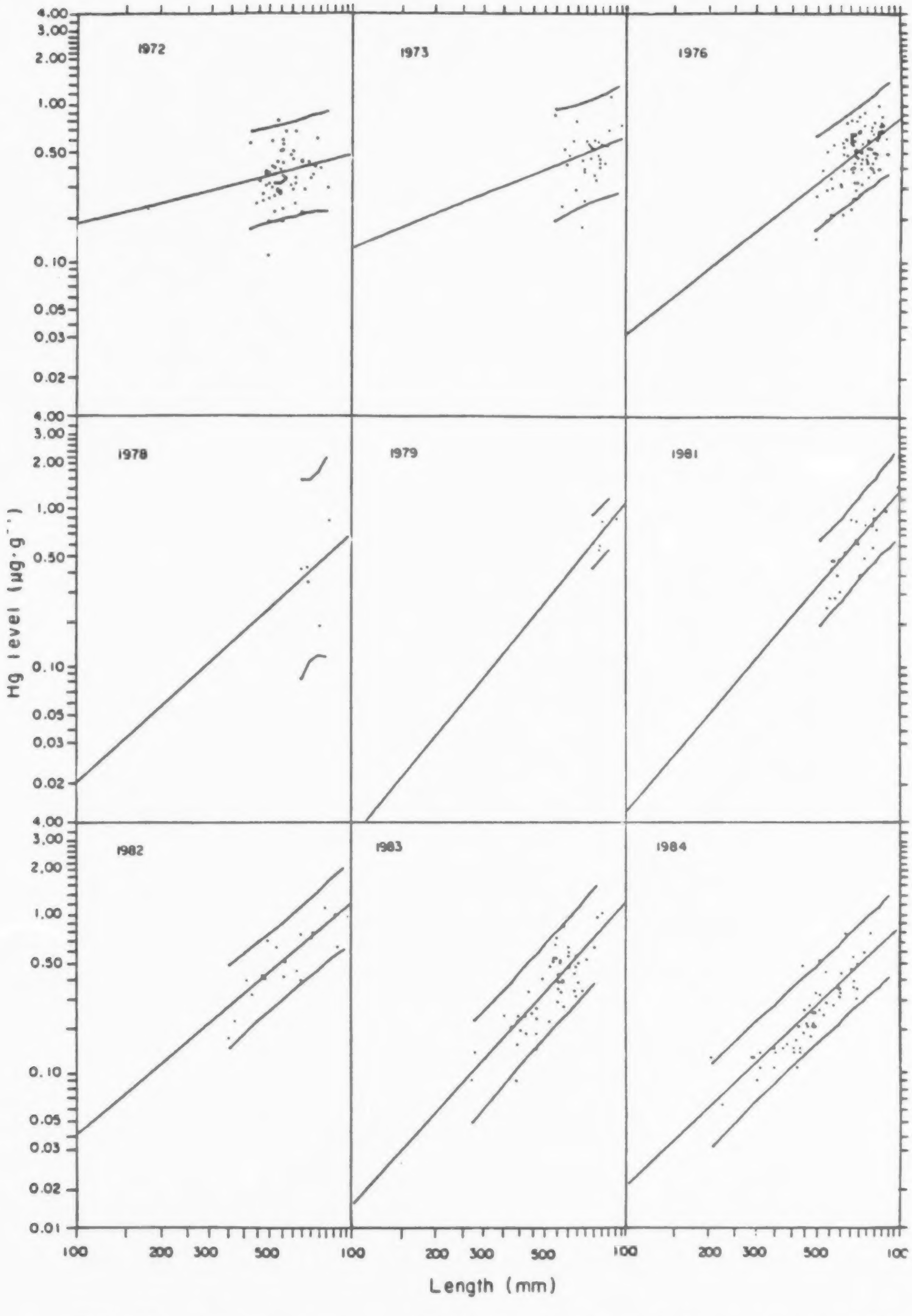

Fig. 30. Log - log regression lines and 95% confidence intervals for muscle mercury vs fork length relationships for northern pike samples taken from Split Lake in 1972, 1973, 1976, 1978, 1979 and 1981 - 1984.

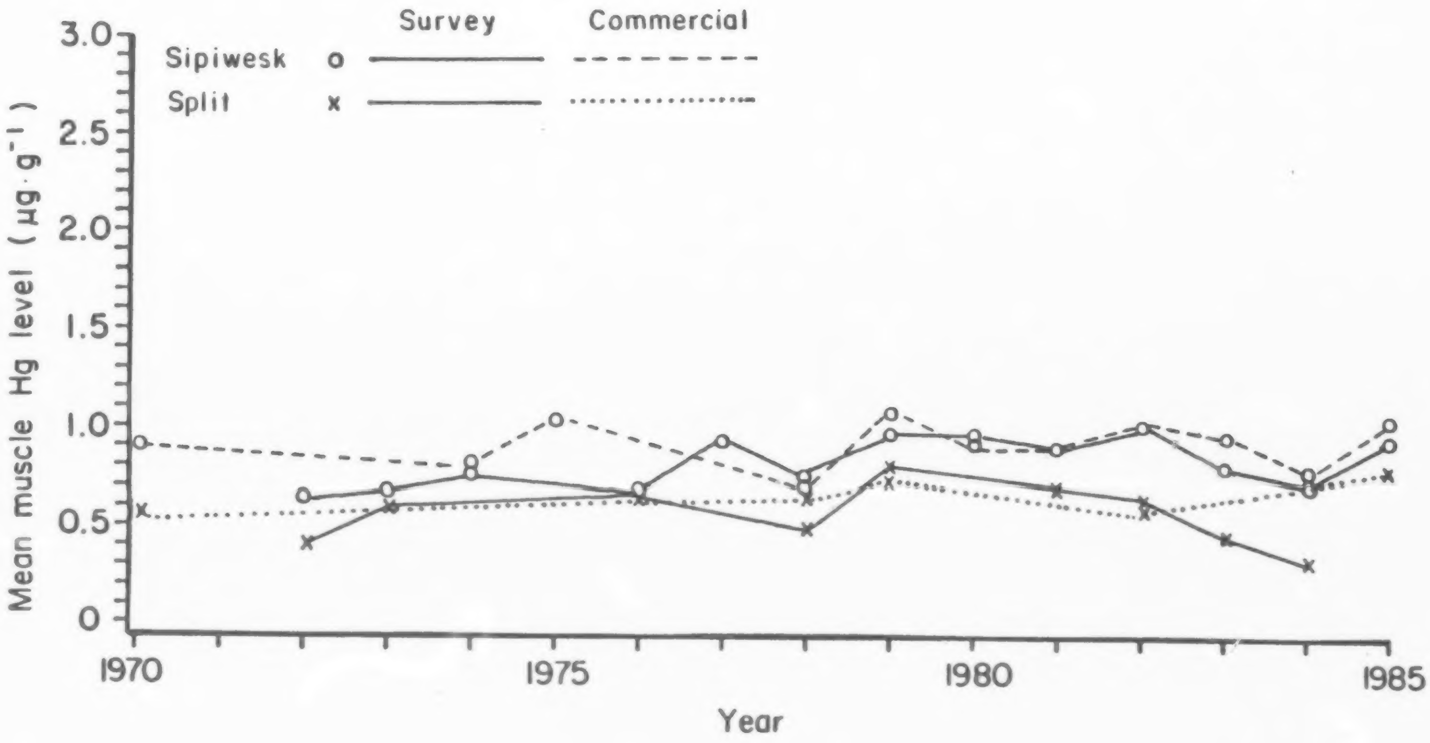

Fig. 31. Annual mean muscle mercury concentrations in survey and commercial samples of northern pike from Sipiwesk and Split lakes over the period 1970 - 1985.

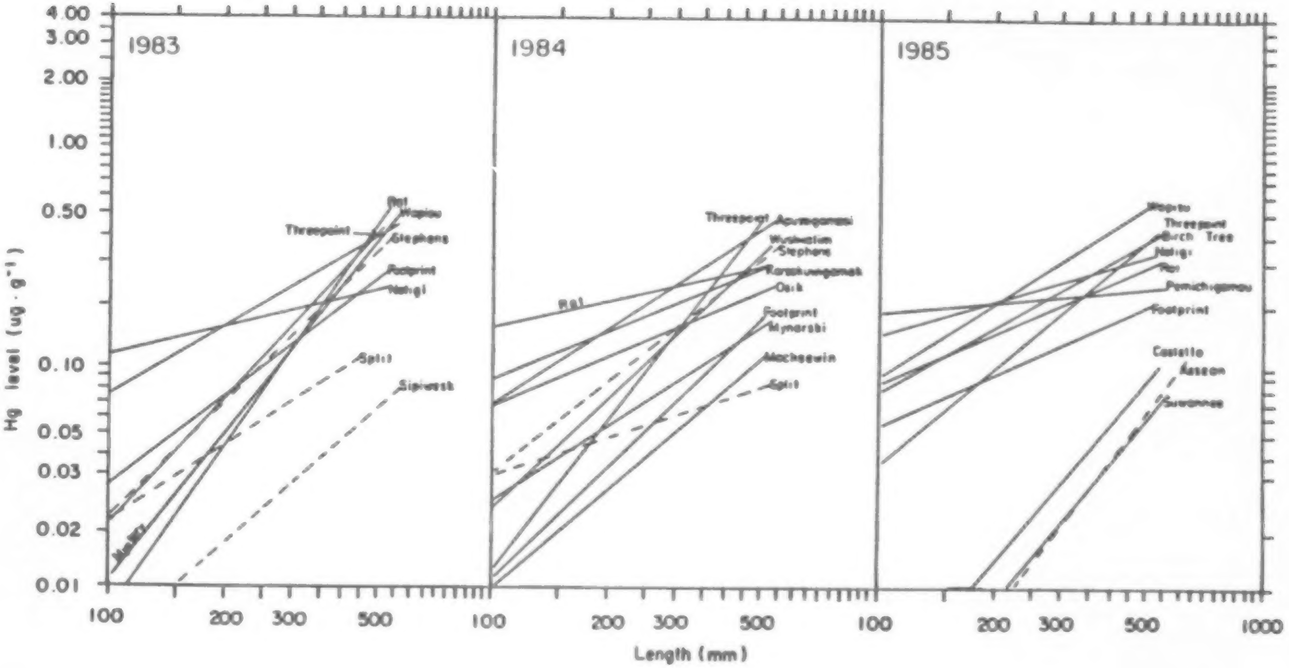

Fig. 32. Log - log regression lines of the muscle mercury vs fork length relationships for whitefish samples taken from the Churchill River Diversion and several Nelson River lakes during 1983 - 1985.

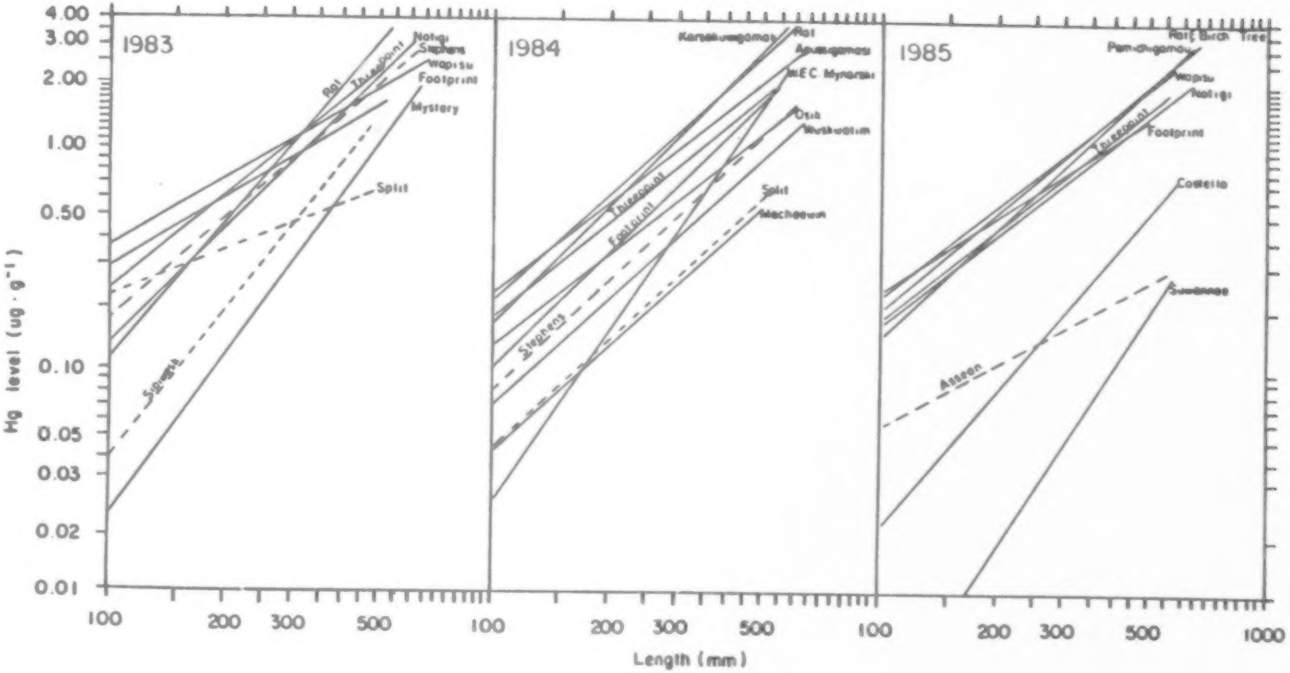

Fig. 33. Log - log regression lines of the muscle mercury vs fork length relationships for walleye samples taken from the Churchill River Diversion and several Nelson River lakes during 1983 - 1985.

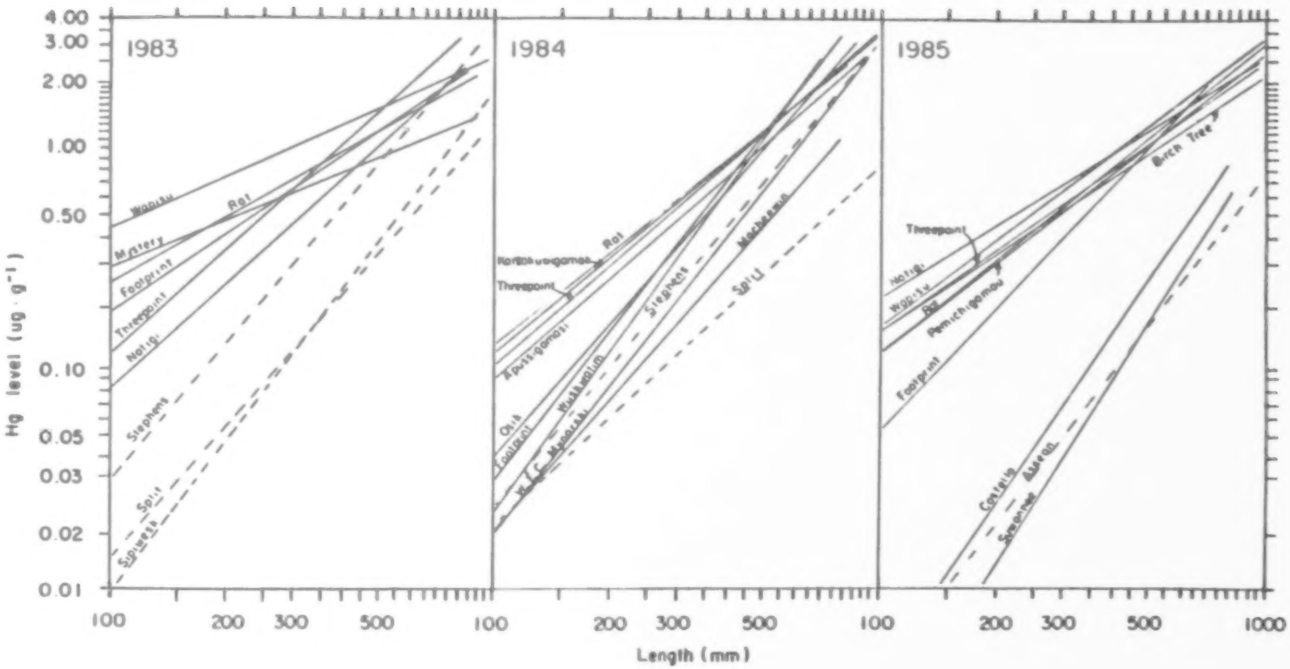

Fig. 34. Log - log regression lines of the muscle mercury vs fork length relationships for northern pike samples taken from the Churchill River Diversion and several Nelson River lakes during 1983 - 1985.

whitefish, walleye and northern pike, as standardized by regression analysis, from lakes along the Churchill River diversion route revealed a progressive downstream decrease in mercury levels in walleye as far as Wuskwatim Lake. The mean mercury level dropped from approximately 2 $\mu g \cdot g^{-1}$ in walleyes from Karsakuwigamak Lake at the upper end of the Diversion to just less than 0.7 $\mu g \cdot g^{-1}$ in Wuskwatim Lake walleye (Fig. 35). This downstream trend was not evident, however, in the mercury concentrations in pike or whitefish. On the contrary, the mean mercury levels, which varied around 1.25 $\mu g \cdot g^{-1}$, were relatively uniform in pike sampled from the Diversion route lakes as far as Apussigamasi Lake. With a couple of exceptions, whitefish from lakes as far downstream as Apussigamasi Lake also exhibited uniform mercury levels which varied around 0.12 $\mu g \cdot g^{-1}$. Pike and whitefish from Wapisu Lake possessed mean mercury concentrations which were slightly higher than mercury concentrations for samples from both upstream and downstream lakes. The pattern of mean mercury concentrations in whitefish samples was similar to that for walleye in that the means for Birch Tree and Apussigamasi lakes were higher than those for Diversion route lakes immediately upstream. Mean mercury concentrations in all three species were lowest in Split Lake, but increased again in downstream Stephens Lake.

The pre - and post - impoundment areas and percentage increases in lake surface areas due to flooding for lakes on the Churchill River Diversion and the Nelson River are presented in Table 10. The greatest percentage flooding was witnessed on Notigi Lake on the Churchill River Diversion immediately above the Notigi Control Dam. The surface area of

Table 10. Comparison of lake surface area increases and fish muscle mercury levels for the Churchill River Diversion and Nelson River lakes. Mercury levels are based on the standard sized fish for each species and respective regression analysis for the most recent year's data.

Lake					Mercury level ($\mu g \cdot g^{-1}$)			
Name+	Area (ha)				Most recent years data	(350 mm) Whitefish	(400 mm) Walleye	(550 mm) Northern pike
	Pre-impoundment	Post-impoundment	Hectare increase	% increase				
Karsakuwigamak	1,788	3,868	2,080	116	1984	0.22	1.96	1.41
Pemichigamau	2,018	3,937	1,919	95	1985	0.22	1.47	1.19
Mynarski*	1,963	3,631	1,668	85	1984	0.10	0.92	0.80
Costello**	1,495	n/a	n/a	n/a	1985	0.04	0.37	0.33
Suwannce**	4,945	n/a	n/a	n/a	1985	0.03	0.11	0.21
Rat	7,881	16,405	8,524	108	1985	0.21	1.52	1.34
Notigi	1,742	9,095	7,585	435	1985	0.28	1.19	1.39
Wapisu	4,478	5,921	1,443	32	1985	0.38	1.56	1.53
Macheewin	1,268	1,288	20	2	1984	0.06	0.40	0.55
Osik	2,256	2,607	351	16	1984	0.17	0.91	1.25
Footprint	2,035	2,807	772	38	1985	0.14	1.14	1.27
Threepoint	4,494	6,626	2,132	47	1985	0.23	1.22	1.27
Wuskwatim	4,789	4,907	118	2	1984	0.18	0.68	1.42
Birch Tree	456	627	171	38	1985	0.26	1.51	1.09
Mystery	904	1,166	262	78	1983	0.17	0.65	0.95
Apussigamasi	1,433	2,235	802	64	1984	0.28	1.53	1.25
Sipiwesk		34,570[1]		20[2]	1983	0.04	0.87	0.48
Split	27,939	27,960	21	<1	1984	0.07	0.43	0.31
Stephens	8,882	29,898[3]	21,016	237	1984	0.19	0.81	0.87

+ Lakes are arranged in upstream to downstream order.
* Mynarski Lake refers to West/Central Mynarski.
** not flooded.
1 from Inland Waters Directorate Environment Canada.
2 pers. comm., Dennis Windsor, Manitoba Hydro.
3 includes old Moose Lake Basin plus old Nelson River channel presently within Stephens Lake Boundaries.

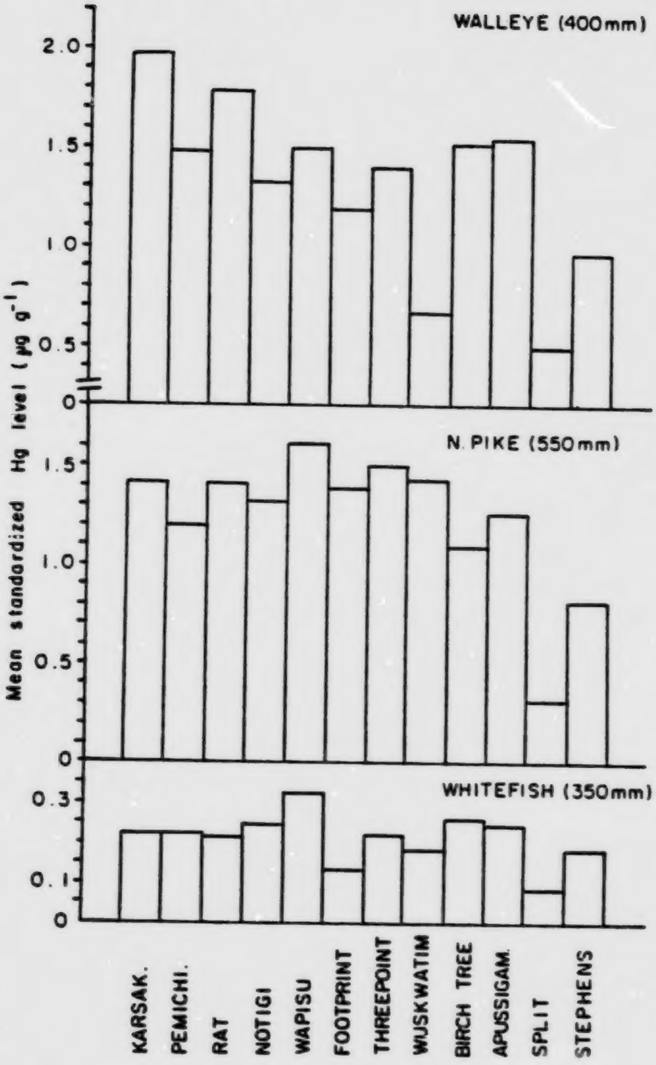

Fig. 35. Average muscle mercury concentrations adjusted to standard lengths for walleye , northern pike and whitefish from lakes on the Rat - Burntwood system and the Nelson River which are influenced by Churchill River Diversion flows. Averages are based on the 1983 - 1985 data.

Notigi Lake increased by 435 per cent or 7585 ha; this meant a quadrupling in size. The greatest absolute area of flooding in the study area, however, was in the forebay of the Kettle Rapids Dam, i.e. Stephens Lake. Here the water surface area increased by just over 21,000 ha or 237 per cent. Much of this increase was due to lateral flooding along the Nelson River. Major flooding (i.e. >1000 ha) occurred on all lakes affected by the Churchill River Diversion within the Notigi forebay. Here the surface area of all lakes, except Notigi Lake, virtually doubled in size after the Diversion was put into operation. The extent of flooding on the Churchill River Diversion downstream of the Notigi Control was not as marked. Although substantial areas of terrain were flooded around Threepoint Lake and Wapisu Lake, the increased lake surface areas were still less than 50 per cent (Table 10). Wapisu Lake presents an interesting situation. Even though the surface area increased by 32 per cent, this flooding is not readily apparent. This is because much of the shoreline around Wapisu Lake is steep and bedrock controlled. The least amount of flooding on any of the lakes directly on the Diversion was on Split Lake. Although the mean annual surface elevation of Split Lake has increased by approximately 1 m, this increase has been contained within the historical range of surface elevation fluctuations on the lake (R. J. Bowering, pers. comm.). There has been only minor flooding on Macheewin and Wuskwatim lakes. Since Macheewin Lake is at the headwaters of the Footprint River, backflooding there has been minimal. Although little of the immediate periphery of Wuskwatim Lake has been flooded, there has been extensive flooding along the Burntwood River immediately upstream of the lake.

No direct relationship between the extent of flooding, either in absolute terms or on a percentage basis, and the level of mercury contamination in whitefish, walleye or northern pike was apparent. However, when both the level of mercury contamination and the degree of flooding were ranked[1] and regression analysis applied to these ranked values, significant rank correlations between flooding and mercury contamination were obtained in the case of all three species (Fig. 36). The best rank correlations were obtained using the exponential function. Although rank correlation indicated a direct relationship whereby the level of mercury contamination in fish from a lake increased with an increase in the area flooded around the lake, it was not a close or well-defined relationship as suggested by the number of outlying points. For example, the point for Wapisu Lake was well removed from the regression curves. In terms of flooding, Wapisu Lake was the fifth lowest, but mercury levels in whitefish (Fig. 36A) and northern pike (Fig. 36C) were the highest, while Wapisu walleye (Fig. 36B) exhibited the second highest mean mercury levels of all lakes examined during the present study. Whitefish and walleye from Birch Tree and Apussigamasi lakes also exhibited an unusually high mercury content relative to the extent to which they had been flooded. The mean mercury concentrations in northern pike from Wuskwatim Lake was the second highest (rank 16), but the lake was one of the least flooded (Fig. 36C). In contrast to these situations, walleye samples from Mystery, Stephens and Notigi lakes (Fig. 36B) and pike from Stephens Lake (Fig. 36C) showed low levels of mercury contamination despite extensive flooding. In fact, Notigi and Stephens lakes, on a percentage basis, were the two most extensively

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[1] Increasing ranks indicate increased mercury levels and flooding.

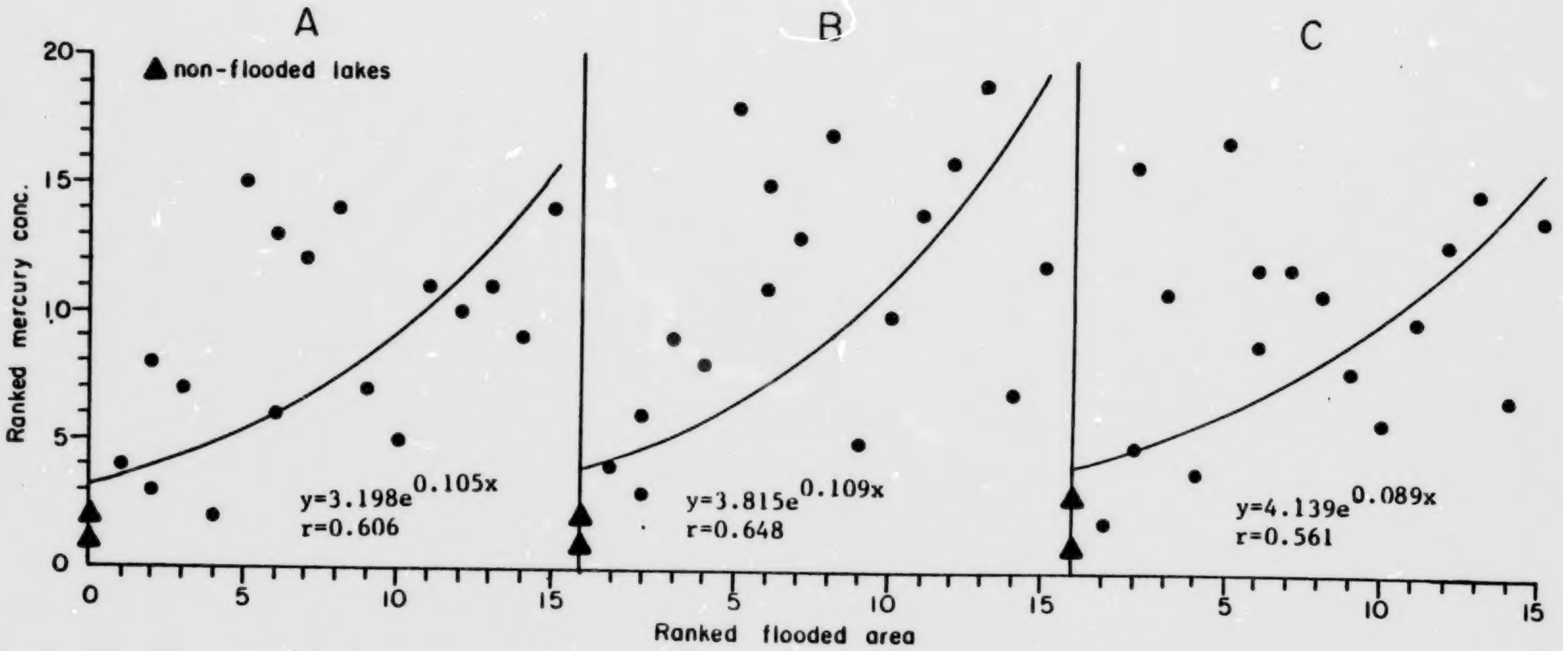

Fig. 36. Relationship of ranked flooded area (percentage lake surface area increase) to ranked, length corrected, mercury concentrations in whitefish (A), walleye (B), and northern pike (C) for lakes on the Churchill River Diversion and Nelson River. Non-flooded lakes are Costello and Suwannee, which are tributary to Rat Lake. The equations for the exponential curves, and correlation coefficients are given in each panel.

flooded lakes. A variety of factors other than flooding, which can influence the availability of mercury and its uptake by fish and which were not examined in this study, may have accounted for these divergent cases. Considering there are relatively few such discrepancies, it is evident that the level of mercury contamination in fishes from lakes on the Churchill River Diversion and the Nelson River is proportionately related to the amount of flooding on these river systems.

The effect of flooding on the elevation of mercury contamination in fish is further supported by comparisons of the mean and range in mercury levels in whitefish, walleye and northern pike for flooded lakes to those for non-flooded lakes within a specific drainage basin. These comparisons are illustrated in Fig. 37 for nine major drainage basins which encompass or lie immediately adjacent to the study area as shown in Fig. 1. Mercury concentrations depicted in Fig. 37 are based on means adjusted for length by the coefficient method. Non-flooded lakes from which fish mercury concentrations were used for comparison are numbered in Fig. 2, and the corresponding names are listed in Appendix II. Although there was relatively little data on whitefish with which to make a comparison, the information did indicate that the average mercury concentrations were higher in flooded than in non-flooded lakes within similar drainage basins (Fig. 37A). In basins 3 and 8, which encompassed the Rat - Burntwood River system and the Split Lake to Stephens Lake reach of the Nelson River, respectively, the mean mercury content of whitefish in non-flooded lakes was less than 0.1 $\mu g \cdot g^{-1}$. The average mercury concentrations in whitefish from flooded lakes were more than double the concentrations noted in the non-flooded lake in

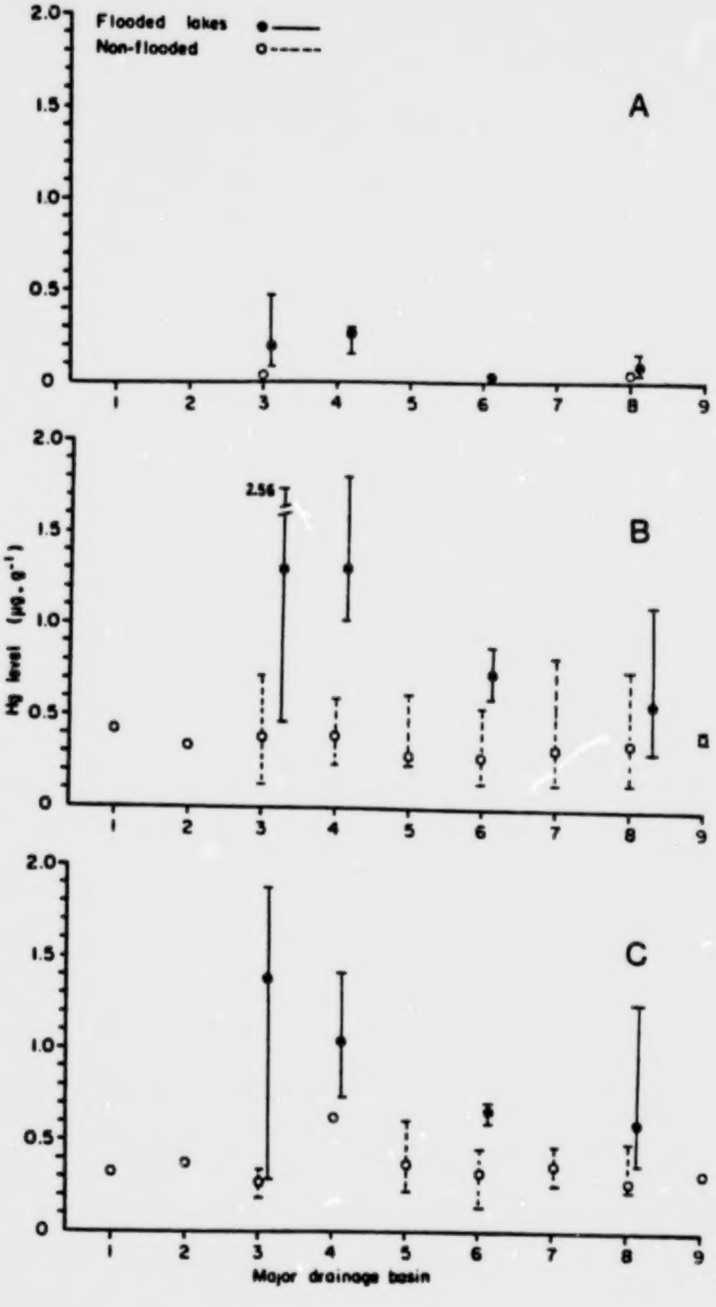

Fig. 37. Averages and ranges in muscle mercury concentrations in whitefish (A), walleye (B), and northern pike (C) from flooded and non-flooded lakes in the major drainage basins as shown in Fig. 2. Mercury concentrations have been corrected to standard sizes.

basin 3. This was not the case, however, for basin 8, where only Stephens Lake whitefish showed any marked accumulation of mercury (Fig. 38). Whitefish mercury levels in Split Lake appear comparable to those of whitefish in Assean Lake which is unimpacted by flows from the Churchill River Diversion. Low whitefish mercury levels are also noted in basin 6, i.e. Sipiwesk Lake.

The average mercury concentrations found in walleye (Fig. 37B) and northern pike (Fig. 37C) in unimpacted or non-flooded lakes both within and adjacent to the study area are generally less than 0.40 $\mu g.g^{-1}$. Although there was some overlap in ranges, the mean mercury concentrations in walleye and pike from flooded lakes greatly exceeded the natural or "background" levels observed in the area. The greatest increases in mercury contamination due to flooding were apparent in basins 3 and 4, i.e. the Churchill River Diversion route from its origin to Split Lake. These increases did not appear to be related to any higher background contamination levels; the mean mercury concentrations in walleye and pike from non-flooded lakes in basins 3 and 4 were not appreciably different from the means for non-flooded lakes in any other basins. As in whitefish, the mean levels of mercury contamination in walleye and pike from basins 6 and 8, the Nelson River lakes, were lower than those for basins 3 and 4, the Churchill River Diversion lakes.

A summary of the liver - muscle mercury ratios found in whitefish, walleye and northern pike from three lakes on the Rat - Burntwood river system (Rat, Footprint and Threepoint) and from Split Lake are presented in Table 11. This information is based on the data from Appendix 19 of Green's (1986) report. Although the ratios were

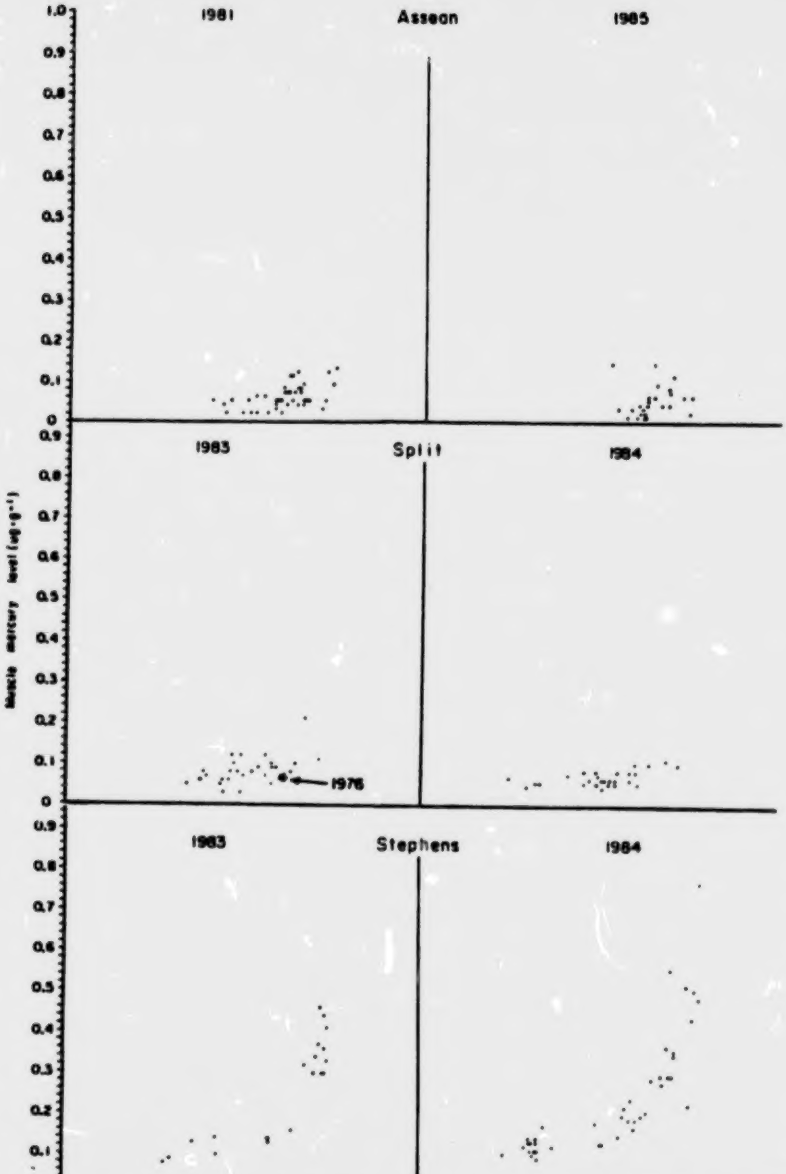

Fig. 38. Comparison of muscle mercury concentrations in individual whitefish sampled from Assean, Split and Stephens Lake between 1981 and 1985.

Table 11. Average and range in liver-muscle mercury ratios for small and large whitefish, walleyes and northern pike from lakes on the Churchill River Diversion. Numbers in parentheses are number of ratios the means are based upon.

	Liver/Muscle Mercury Ratio in -			
	Rat L.	Footprint L.	Threepoint L.	Split L.
Lake Whitefish				
< 300 mm FL				
Mean	2.06 (4)	2.48 (4)	2.08 (1)	0.89 (3)
Range	1.90-2.11	2.00-3.23	-	0.50-1.57
> 300 mm FL				
Mean	2.01 (3)	1.74 (7)	1.82 (6)	1.39 (7)
Range	1.55-2.47	1.11-2.62	1.68-2.08	1.25-1.55
Walleye				
< 300 mm FL				
Mean	0.91 (3)	0.84 (4)	0.54 (3)	0.61 (3)
Range	0.77-1.69	0.72-0.97	0.44-0.69	0.47-0.73
> 300 mm FL				
Mean	1.37 (3)	0.74 (11)	0.80 (6)	0.69 (4)
Range	1.06-1.90	0.46-1.34	0.42-1.06	0.55-1.09
Northern Pike				
< 400 mm FL				
Mean	1.47 (1)	0.92 (2)	1.10 (2)	0.86 (1)
Range	-	0.73-1.11	-	-
> 400 mm FL				
Mean	1.64 (5)	1.19 (11)	1.25 (6)	0.70 (4)
Range	1.36-2.19	0.73-1.65	0.84-1.50	0.59-0.78

highly variable, there was a trend for the liver - muscle ratio to decrease with increasing size in whitefish and increase with increasing size in walleye and pike from the Rat - Burntwood lakes. In the case of Split Lake whitefish, however, there was a pronounced increase in the liver - muscle mercury ratio with size. For walleye and pike from Split Lake, the liver - muscle mercury ratios appeared to remain stable or perhaps decrease slightly with size.

DISCUSSION

Since the pioneer research into environmental mercury contamination in Sweden during the latter part of the 1960's and the subsequent general recognition of the problem in North America in 1970, numerous studies have been conducted on mercury pollution. Most of these studies have been on the behaviour of mercury, primarily from anthropogenic sources, in the aquatic environment and particularly on the accumulation of mercury by fish. Within the past decade, it has also been recognized that the creation of reservoirs may result in the elevation of mercury concentrations in fish. In Appendix IV, reference is made to the authors of a number of studies listed under several subject areas related specifically to mercury in the aquatic environment. Although these listings by no means represent an exhaustive bibliography of the available literature on mercury pollution, the papers do provide a comprehensive overview of the status of current knowledge on mercury in the freshwater environment. We do not intend to fully review these previous investigations in this report, since many have been reviewed by others (e.g., Krenkel 1973, 1974 and Jones *et al*. 1986) or are considered by other study sectors of the present Agreement. Although the references in Appendix IV may or may not have any direct relevance to the present study, they contain information which provided further insight to one of the problems at hand; i.e., to account for elevated mercury concentrations in fishes in the Churchill River Diversion and Nelson River lakes.

As expected, elevated mercury concentrations in fishes from the study area were evident primarily in the large piscivorous species; i.e., northern pike, walleye and sauger (Table 4). Sauger exhibited higher mean concentrations than pike or walleye even though the two latter species contained mercury levels in excess of 4.0 $ug.g^{-1}$. The other fish species, which are either benthic, planktonic or omnivorous feeders, did not contain excessively high mercury levels. Previous investigators (Jernelov and Lann 1971, Jernelov 1972 b, Fagerstrom and Assell 1973, Potter et al. 1975, etc.) have attributed the higher mercury content of large piscivores to the bio-magnification of mercury at successively higher trophic levels in the aquatic food chain. It has also been observed that feeding and behavioural differences between species may account for the differences in mercury levels in the various species, even those with very similar piscivorous habits. Mathers and Johansen (1985) attributed the fact that walleye had higher mercury concentrations than pike to the high food consumption rate, lower food conversion efficiency and greater activity level of the walleye . More recently it has been suggested that the physiology of a fish species may affect its ability to accumulate mercury. Jackson (1986) speculated that the lake whitefish may have a peculiar biochemical mechanism for ridding itself of methylmercury. Support is given to this hypothesis by the observations of MacCrimmon et al. (1983), Mathers and Johansen (1985) and Skurdal et al. (1985). MacCrimmon et al. (1983) found that the pronounced rapid accumulation of mercury in lake trout from nine Ontario lakes was directly correlated to the consumption of smelt (*Osmerus mordax*) and the mercury concentrations in those smelt. A similar relationship was found by Mathers and Johansen (1985) between walleye and pike mercury levels and smelt consumption in Lake Simcoe, Ontario. Skurdal *et al*. (1985), in contrast, found that in Arctic charr (*Salvelinus alpinus*), which also foraged on smelt (*O.eperlanus*), the accumulation of mercury was slower than in other predatory fish such as large brown trout (*Salmo trutta*) and approximately the same as in the planktivores (i.e., whitefish, *Coregonus lavaretus* and smelt). Results on liver-muscle mercury ratios from the present study also suggest that the whitefish may be better able than walleye or pike to rid itself of mercury. The liver-muscle mercury ratio in whitefish tended to decrease with size or age (Table 11); this suggests decreasing mercury concentrations in the liver. Liver-muscle mercury ratios in walleye and pike, on the other hand, increased with size. Since, as indicated by McFarlane and Franzin (1980), the liver removes mercury by sequestration to form metallothioneins, the higher liver mercury concentrations relative to mercury concentrations in the muscles of younger whitefish may represent an elimination process which is proceeding at a faster rate than the rate of accumulation.

In the present study, fish age was found to be most often positively and strongly correlated with mercury concentrations in whitefish and walleye, but not in northern pike (Table 5). This contrasts with the results of Scott (1974) who consistently found "extremely" significant positive relationships between age and mercury content in northern pike, as well as in whitefish and walleye. In our case, there was a stronger relationship between length and mercury in pike. This agrees with Olsson's (1976) study on pike, which found the best correlation to be between mercury content and length. The reasons for these discrepancies are not readily apparent, but may be attributable to differences in ageing techniques. Scott (1974) used scales to age pike, whereas we used cleithra and fin ray sections. Annular markings on these structures are easier to distinguish and generally reveal older age groups than do pike scales. There is usually a considerable overlap in the age-length distributions for pike within and particularly between year classes (Derksen and Gillies, in prep.). Johnels et al. (1967) observed some relationship between mercury levels and age of pike, but dismissed using this relationship in their study because of unreliable age determinations from pike scales. Length therefore may provide a better correlate with mercury content in pike, since it reflects both metabolic growth and age, which as indicated by Scott (1974), respectively determines the physiological and chronological exposure times of the fish to a contaminant. In walleye , age provided only a slightly better correlation with mercury content than length. As suggested by Skurdal et al. (1985), this is probably due to the marked reduction in growth once sexual maturity is attained. Walleye, particularly in moderately fished or non-exploited northern lakes such as those in the present study area may survive upwards of 20 years and show virtually no growth following sexual maturation, which may occur after

8-10 years (Lysack 1982). Older walleye, because of their longer exposure, may therefore exhibit higher mercury concentrations than younger walleye of similar size. Although weight, relative to age, was poorly correlated with mercury content in whitefish, it was still better than length. This is because whitefish reach their maximum growth in length relatively rapidly, but continue to put on weight.

The relationship between fish size or age and mercury uptake appears to be primarily exponential in nature (Table 6). The only exceptions were the relationships of mercury in whitefish to age and weight; these tended to be linear. The exponential character of the mercury versus size or age relationships is due to the much lower mercury concentrations in the smaller, younger fish. This situation seems anomalous since it might be expected that mercury concentrations in the younger, faster growing fish, because of their higher metabolic rate and greater effective surface area for absorption of mercury, would be higher than or at least similar to concentrations in the older, slower growing fish of a given species. Kudo (1976) noted in an experiment using radioactive tracers that younger fish (guppies) generally accumulated mercury desorbed from bed sediments at a faster rate and to higher concentrations. Because of the experimental conditions, this may only be a short term phenomenon. Under natural conditions, the higher metabolic rate of young/small fish may, in fact, have the opposite effect and serve to increase the clearance of mercury from their tissues. de Freitas _et_ _al_. (1974) observed that the mercury clearance rates in walleye and sauger, although inversely related to weight, were directly related to metabolic rate. Sharpe _et al_. (1977) made a similar observation on goldfish under experimental conditions. Methylmercury was cleared from the tissues of small goldfish at a faster rate per unit weight of body tissue than from larger speciments. Clearance rates may also depend on the type of mercury present in fish. Bache _et al_. (1971) found that less than 50 per cent of the total mercury in young lake trout (less than 3 years of age) consists of methylmercury, with the remainder being in inorganic form. It is known that inorganic mercury is cleared from living tissue much faster than organic or methyl mercury. This, coupled with increased clearance rates at higher metabolic rates might therefore account for the lower mercury concentrations in young or small fish. This hypothesis is contingent on the assumption that the young of all species contain proportionately low levels of methylmercury. This assumption may not necessarily be valid; Rodgers and Qadri (1982) reported that 68 to 83 per cent of the total mercury in yearling yellow perch (_Perca flavescens_) was methylmercury. The lower mercury levels in the younger fish may also be simply a function of the mercury content in their diet. If the younger fish were exposed to food with higher mercury content, they would accumulate higher concentrations. Scherer _et al_. (1976) experimentally exposed yearling walleye to a highly contaminated diet and noted that they accumulated much higher levels than the yearlings in the controls which had been fed a relatively uncontaminated diet. In studies on feral fish, MacCrimmon _et al_. (1983) found that tissue mercury concentrations increased sharply in lake trout once they began feeding on smelt which contained higher mercury levels. Skurdal _et al_. (1985) reported a similar change in the mercury content of brown trout after they passed from the parr to adult stage, moved from the river into a lake and changed their diet from insect larvae to fish. Mathers and Johansen (1985) also noted that the mercury concentrations in the prey of yearling walleye was less than in the prey of older walleye. Finally, and perhaps most importantly, the higher levels in the older fish as opposed to lower levels in the younger fish are obviously a direct consequence of a longer exposure to elevated environmental mercury levels (Scott 1974).

The mercury levels in fish at a given age or size is the result of the dynamic relationship between mercury uptake and clearance rates. Uptake rates are basically a function of the absorption of mercury in the water via the gills and/or of mercury in food via the fish's intestinal tract (Fagerstrom _et al_. 1974). A wide variety of complex factors may influence mercury uptake in fish. These factors may be grouped into primary categories such as species and diet (Mathers and Johansen 1985), temperature (Reinert _et al_. 1974, Rodgers and Beamish, 1981, and Bodaly _et al_. 1986), water chemistry and general trophic or nutrient conditions (eg., Hakanson 1980, Wright and Hamilton 1982 and Rudd and Turner 1983). Water quality conditions which are particularly important in determining mercury uptake are pH (Kleinert 1972, Hakanson 1980 and Wren and MacCrimmon 1983), calcium (Rodgers and Beamish 1983) and selenium

(Phillips and Buhler 1978, Rudd et al. 1980 and Speyer 1980). With respect to clearance rates, they may be the result of excretion and/or growth dilution. Since most mercury in fish appears to be in the methylated form and since methylmercury binds strongly to the sulphydryl molecule in proteins, clearance rates of methylmercury from fish muscle tissue may be extremely slow. Lockhart et al. (1972) and Uthe (1972) have determined that the half-time of mercury in northern pike may be as much as two years. Despite the growing number of studies in recent years on mercury pathways in freshwater environments and aquatic food chains, as well as mercury kinetics in fish, there is still considerable disagreement in the literature over whether the water or food route is more important in determining contaminant levels in fish (Murty 1986). No conclusive evidence was generated by the present study which could assist in resolving this fundamental question on mercury uptake.

In environments where ambient mercury is absent or extremely low, an equilibrium appears to exist between mercury uptake and clearance in fish; as rapidly as fish take up mercury, it is excreted or diluted by the increased body mass. In this situation, there is no accumulation of mercury with time or increasing fish size. This has in fact been documented by Johnels et al. (1967). They found that pike from a sparsely farmed, uncontaminated area in Sweden possessed a constant muscle mercury level of approximately 0.10 $\mu g.g^{-1}$ even though the fish ranged in weight from 0.5 to 3.0 kg. Fish which contain less than 0.20 $\mu g.g^{-1}$ mercury do not exhibit an increasing concentration with size or age, are generally viewed as being uncontaminated (Johnels et al. 1967, Kent and Johnson 1979). Where the environmental mercury levels are sufficiently high enough for uptake to exceed clearance rates, mercury will slowly accumulate with increasing fish size. This slow accumulation of mercury is illustrated by the mercury concentrations in a sample of 10 northern pike taken from Big Whiteshell Lake, a Precambrian Shield lake in southeastern Manitoba, on March 29, 1972 (Fig. 39). Even though pike in this sample were as large as 8 to 9 kg, mercury concentrations did not accumulate beyond 0.40 $\mu g.g^{-1}$. This slow accumulation contrasts sharply to that observed for pike taken in the present study from flooded lakes on the upper end of the Churchill River Diversion. Here pike in the 2 to 3 kg size range contained an average of 1.5 to 2.0 $\mu g.g^{-1}$ mercury, or 10 times as much as pike of an equivalent size in Big Whiteshell Lake. Some fish species in some of the lakes within the present study area, however, did illustrate very low accumulations of mercury. These were apparent in whitefish from Costello, Suwannee, Sipiwesk, Split and Assean lakes, in walleye from Suwannee and Assean lakes in 1985 and in northern pike from Suwannee Lake (Green 1986). In these cases, mercury in whitefish did not accumulate to concentrations in excess of 0.20 $\mu g.g^{-1}$ in any individual, while in walleye and northern pike concentrations in individual fish were generally less than 0.30 or 0.40 $\mu g.g^{-1}$. The whitefish, walleye and pike in these cases may be considered to be only slightly or moderately contaminated and to reflect "background" mercury levels in the study area. These

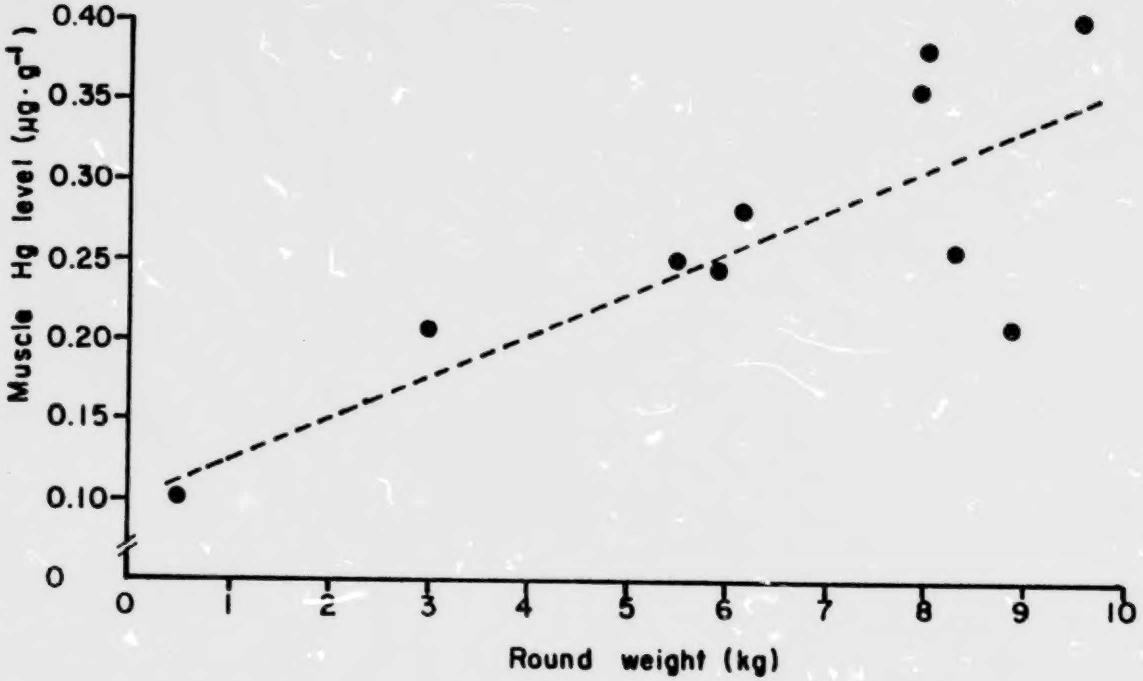

Fig. 39. Muscle mercury levels in a sample of 10 northern pike taken from Big Whiteshell Lake, Manitoba, on March 29, 1972. The dashed line is eye-fitted.

"background" levels also appear to be higher than "background" levels in the adjoining area in Saskatchewan. Here, Murray (1978) reported average muscle mercury concentrations of 0.05 $\mu g.g^{-1}$ in whitefish and approximately 0.28 $\mu g.g^{-1}$ in walleye and northern pike.

Regression analysis of the relationships between mercury content and fish size in the present study suggested that accumulation rates for whitefish, walleye and northern pike, although species dependent, were largely constant. Results of covariance analysis more frequently indicated differences in regression line elevations than in slopes (Tables 7, 8 and 9). In cases when significant differences were found in slopes, these differences were often due to either inadequate sample size or a limited size range in one of the samples. The tendency for mercury versus size regressions to be consistent in terms of slope supports the suggestion that the rate of mercury accumulation by fish within the study area, and particularly the Notigi forebay, was generally fixed for each species. The concept of a species-specific fixed rate of mercury accumulation is also suggested by the results of recent laboratory and field studies on mercury uptake. Rodgers and Beamish (1981), examining the uptake of methylmercury in rainbow trout at temperatures of 10°C and 20°C, observed a constant ratio of uptake efficiency of methylmercury relative to oxygen and surmised that this ratio (approximately 0.25) was characteristic of the compound and the experimental conditions. This study considered mercury uptake from water. Consistency in mercury uptake also appears to apply when the source of mercury is food. Mathers and Johansen (1985) found that mercury accumulation rates, which were related to food consumption rates and food conversion efficiencies, did not change for walleye and northern pike as they matured. The problem of whether most of the mercury in fish enters via the gills or the gastro-intestinal tract is perhaps immaterial in determining the ultimate mercury accumulation rate, since Phillips and Buhler (1978) found that mercury from both sources is simply additive. If mercury accumulation rates are in fact fixed for a given species under given environmental conditions, then it follows that any increase or decrease in the environmental mercury level would be reflected in a corresponding increase or decrease in the elevation of the mercury versus size regression line. Statistically significant differences in regression line elevations were recorded in 50 to 60 per cent of the annual comparisons made for whitefish, walleye and northern pike (Tables 7, 8 and 9). The biological significance of these results is uncertain, however, since most of the comparisons involved data from fish samples collected before 1983. As such, these samples and even some taken during the present study were largely represented by insufficient numbers of fish or an inadequate size range. Where comparisons were made based on the more complete 1983 - 1985 sample data, significant differences in the elevation of the mercury versus size regressions were observed in 3 out of 7 comparisons for whitefish, 4 out of 8 for pike and 5 out of 7 for walleye. Closer scrutiny of the data involved in the comparisons showing statistically significant differences, however, again indicated that sampling error (i.e., inadequate size range, sample size or age representation) may have been responsible for these differences. Where the comparisons were based on adequate samples, they indicated that there may have been slight decreases in the rates of mercury accumulation between 1983 and 1985. These decreases appeared to be limited to waters downstream of the Notigi Control Dam.

Log-log regression lines fitted to the 1983 - 1985 data for the most part appeared to represent mercury accumulation rates for the whitefish, walleye and northern pike. With few exceptions, the 95 per cent confidence intervals about these regression lines encompassed most of the respective data points for samples collected prior to 1983. Between year comparisons of the sample data for each lake indicated that there have been minor, if any, temporal changes in the rates of mercury accumulation or mercury levels in the three species within the study area over the period of available mercury records. The absence of any significant downward trend in mercury accumulation was particularly apparent in the mercury levels in pike in Split and Sipiwesk lakes (Fig. 31) for which sample records covered a 16 year period.

The between lake comparisons of the log-log regression lines (Fig. 32 - 34), although exhibiting some annual variability, revealed that the regression line slopes were generally similar while the elevation of the lines varied from lake to lake. This observation was particularly applicable for lakes lying on the Churchill River Diversion and especially for those in the Notigi reservoir. The similarity in slopes indicates that the rates of mercury accumultion are basically same, but that the ambient level of mercury that is available for uptake in each lake, as reflected by the regression line elevations, can vary considerably. Based on these comparisons, it is evident that the lakes lying directly on the Diversion in the Notigi Reservoir and immediately downstream of the Notigi Control Dam are highly contaminated with mercury.

Highest mercury levels appeared to occur in Karsakuwigamak, Pemichigamau, Rat and Notigi lakes. The elevated mercury levels in these lakes have been related to the extensive flooding in the area following closure of the Notigi Control Dam (Bodaly _et al_. 1984). The lakes in the Notigi reservoir, and particularly Rat Lake, were the most severely flooded lakes along the Diversion route. Surface areas of these lakes were increased by an average of 170 per cent (Table 10). Abernathy _et al_. (1985), who determined mercury concentrations in water during the filling of a reservoir, found that the increases in mercury levels at different transects in the reservoir were directly correlated with the increase in surface area at the transect. Mercury levels in Wapisu Lake, which is immediately downstream of the Notigi Control Dam, and Threepoint Lake which lies at the confluence of the Rat, Burntwood and Footprint rivers, appeared to be similar to those in the Notigi reservoir. Flooding on Wapisu and Threepoint lakes, however, was from 2 to 10 times less the flooding on Notigi forebay lakes and mercury levels would therefore have been expected to be correspondingly lower. It should furthermore be noted that flooding effects are not as evident on Wapisu Lake since most of the lake's shoreline is bedrock. Abernathy _et al_. (1985) observed no increase in water mercury concentrations at transects overlying bedrock. By this token, mercury levels in Wapisu Lake should certainly have been lower than was indicated by the fish mercury concentrations.

The higher mercury levels in Wapisu Lake, in most probability, result from the downstream transport of mercury from the Notigi reservoir. High mercury levels have been recorded at the lower end of a reservoir and usually in midstream or the area of greatest flow. These high levels have been attributed to the transport of materials to which mercury was bound (Potter et al. 1975) and increased aeration which enhances the availability of inorganic mercury (Jackson 1986); these factors are directly related to flow conditions. Since mercury tends to bind to smaller particles which remain in suspension longer, mercury from a given source may be transported for a considerable distance downstream (Cranston and Buckley 1972). A similar process may, in part, be responsible for the higher than expected mercury levels in Threepoint Lake. The downstream transport of mercury may have also been responsible for much of the mercury contamination witnessed in Wuskwatim Lake fish.

Mercury levels in Wuskwatim Lake appeared to be similar to those in the Notigi forebay lakes. Flooding on Wuskwatim Lake, however, was very minor; the surface area on the lake increased only 2 per cent (Table 10). A similar amount of flooding occurred on Macheewin Lake, and mercury levels in this lake were the lowest of all lakes on the Churchill River Diversion. Extensive flooding along the banks of the Burntwood River immediately upstream of Wuskwatim Lake, as well as mercury transported from areas even further upstream, may have caused the high mercury levels in Wuskwatim Lake.

The possibility of the downstream transport of mercury contamination is further evidenced by apparent lower mercury levels in lakes along the Churchill River Diversion route which have been backflooded by impoundment and increased flows, but are not directly influenced by diversion flows. For example, the western and central portions of Mynarski Lake, which is located in the Notigi reservoir, were backflooded to almost the same extent (85%) as other lakes in the Notigi reservoir. If the extent of flooding was the sole factor in determining increased mercury contamination in fish, then mercury levels in Mynarski Lake should have approximated those noted for other lakes in the Notigi reservoir. The lower levels in Mynarski Lake could be the result of the fact that the lake is not directly affected by Churchill River flows and consequently downstream mercury transport in the Diversion. The same circumstances appear to apply to Osik and Footprint lakes. Both are located on the Footprint River and have been backflooded by the increased diversion flows. Even though Footprint Lake was flooded to a slightly greater extent than Wapisu Lake, mercury levels in Footprint Lake were still lower than in Wapisu Lake.

Local conditions, such as differences in background mercury levels or water quality, as well as types and amounts of flooded vegetation, in addition to the extent of flooding, may be important factors in determining the level of mercury contamination (Jones et al. 1986). Local conditions may have been the cause of the mercury levels in Osik Lake which were higher than in Footprint Lake, even though the latter was flooded twice as much. Since no data were gathered on physical or limnological conditions within the study area, the influence of such local conditions on our results cannot be assessed.

Another factor which may have significantly influenced results on the level of mercury contamination in fish and which we could not account for is the movement of fish from lake to lake along the Diversion. Increased flows, even in large river systems such as the Rat-Burntwood may stimulate the downstream movement of fish. The extent of movements may be directly proportional to the volume of flows (Derksen and Gillies, in prep.). The fish populations sampled in the Notigi forebay lakes may therefore have been relatively homogeneous. This may account for the relative uniformity in mercury levels in fish from the Notigi forebay. If any downstream fish movements did occur, they were in all probability limited to relatively small numbers of fish or short distances. Such movements would in part explain the high mercury levels observed in northern pike in Wuskwatim Lake. This lake has experienced minor flooding, but there is extensive flooding on the Burntwood River upstream of the lake. Pike, which typically inhabit flooded vegetation or marshy areas, may have moved into Wuskwatim Lake from these upstream areas where they could have acquired high mercury concentrations. It should be noted, however, that most of the fish samples collected in the present study period (1983 - 1985) were taken from late June through early August. Movements of freshwater fishes during this part of the year are generally restricted to relatively limited feeding areas.

Mercury levels in our samples may therefore primarily reflect conditions in the lake from which the samples were taken.

Mercury levels in Birch Tree, Mystery and Apussigamasi lakes in the Thompson area provide an interesting comparison to those observed in the lakes of the Churchill River Diversion. With the exception of Mystery Lake walleye, mercury levels in the three lakes appear very similar to those determined for lakes in the Notigi reservoir (Figs. 32 - 34). The extent of flooding on the Thompson area lakes, however, is considerably less (38 - 78%) than that experienced on the Notigi forebay lakes. It might be speculated that mineralization localized in the Thompson area or fallout of mercury emissions from the nickel smelter at Thompson may in part contribute to the unusually high mercury levels in fish from the area. However, mercury levels in fish from unimpacted or non-flooded lakes in the Thompson area did not differ appreciably from those for other unimpacted lakes throughout the study area and adjoining drainage basins (Fig. 37).

Comparison of log-log regression lines obtained for fish samples from the Nelson River lakes (i.e., Sipiwesk, Split and Stephens) to those for fish from the Churchill River Diversion again indicated that the extent of flooding is not the sole factor which determines the ultimate level of mercury contamination in fish. Even though Stephens Lake was one of the most extensively flooded waterbodies considered in this study, and second only to the flooding on Notigi Lake, mercury levels appeared to be closer to those witnessed in lakes in the Nelson House area than in the Notigi forebay area of the Diversion. Although the exact causes of the lower mercury levels in Stephens Lake are not known, water quality differences between the Churchill River, which is the source of most flows in the Diversion, and the Nelson River may be a factor. For example, calcium ion concentrations in the Rat-Burntwood rivers and the Churchill River are two to three times lower than in the Nelson River at Stephens Lake (Cleugh 1975). High calcium concentrations and alkaline conditions have been found to be coincidental with lower mercury content in fish (Jones et al. 1986, Kleinert 1972). It should also be recognized that Stephens Lake is situated on the geographical boundary between the Precambrian Shield and the Hudson Bay Lowlands. Because of the geological and topographical differences between these areas, Stephens Lake lies in a transitional vegetation area where forest stands begin to decrease (Weir 1983). The reduced level of mercury in Stephens Lake may therefore be partly the result of a decreased forest cover.

Based on the results of the regression analysis, mercury levels in Split Lake appeared to be either slightly higher or equal to those in Sipiwesk Lake (Figs. 32 - 34). This is contrary to what we would have expected based on the negligible flooding on Split Lake and the 20 per cent flooding on Sipiwesk Lake. Mercury levels in Split Lake appeared to be more similar to those in Macheewin Lake, which was also subjected to a very limited amount (2%) of flooding. If the mercury levels are indeed lower in Sipiwesk Lake than in Split Lake, then the difference cannot be attributed to water quality differences in such parameters as calcium. Concentrations of this ion were approximately the same in both Split and Sipiwesk lakes (Cleugh 1975).

Observations drawn from the lake to lake comparisons of length adjusted mercury concentrations for whitefish, walleye and northern pike (Fig. 35) were in some respects similar to those concluded from the comparison of regression lines. As in the regression comparisons, the adjusted mercury concentrations in walleye indicated that the highest levels of contamination occurred in the Notigi forebay lakes and that the levels in the Thompson area lakes (Birch Tree and Apussigamasi) were similar to those in the Notigi reservoir. The adjusted mercury concentrations of fish from Stephens Lake were lower than anticipated given the degree of flooding experienced in the area. As expected, Split Lake, the least flooded of all the impacted lakes sampled, revealed the lowest adjusted mercury levels.

The presence of a decreasing downstream trend in mercury concentrations in walleye is significant. As will be discussed later, this trend was related to the degree of flooding. Similar trends were not evident in the mercury levels in whitefish and northern pike. In contrast to the walleye, both whitefish and pike from lakes along the Diversion exhibited relatively uniform mercury levels (Fig. 35). The uniform mercury levels in pike may be directly attributable to the habitats they typically occupy. Pike spend most of their time in vegetated inshore areas (Inskip 1982). Current investigations have shown that in reservoirs mercury methylation rates are higher in the inshore areas, and particularly in flooded forests, than over the existing, offshore lake sediments (Jackson 1986). Pike, with their predilection for such areas, would therefore be exposed to higher mercury uptake from both the water and their food and this exposure would tend to be consistent from lake to lake irrespective of variations in the degree of flooding. This explanation could in part account for the mercury concentrations in Wuskwatim Lake pike which were the same as those in pike from the Notigi forebay area even though flooding on Wuskwatim Lake was minimal.

The tendency for uniform mercury levels in whitefish may similarly be ascribed to their habitat preference. Adult whitefish prefer deeper, cooler water and are most often found in mid-lake. They will only move into inshore areas in the late autumn when water temperatures are less than 10°C. This behaviour positions whitefish over old existing lake sediments for most of the year where, as indicated by Jackson (1986) and Hecky et al. (1986), mercury methylation rates are lowest. Whitefish movements into inshore areas during autumn also would not expose them to high mercury concentrations in the water since the peak in methylation activity in the sediments occurs in late summer or early autumn (Ramsey and Ramlal 1986). The tendency for whitefish to occupy the water column over existing lake beds, where methylation rates are low, would also account for their generally low level of mercury contamination. However, if whitefish were required to occupy the water column over flooded ground, a higher accumulation of mercury may occur. A more pronounced mercury accumulation was observed in whitefish in Stephens Lake than in Sipiwesk Lake (Green 1986). Because of the extensive flooding on Stephens Lake (237%), whitefish would be more frequently placed over or near flooded ground where higher methylation rates occurred. In Sipiwesk Lake, which was flooded to a much lesser extent, whitefish would tend to be more restricted to waters within the boundaries defined by the preflooded lake basin and consequently to areas of lower ambient mercury levels.

The greater variability in mercury concentrations in walleye may also be attributed to their habitat selection. Unlike pike and whitefish, adult walleye occupy both offshore and inshore areas. They move into littoral areas to feed during periods of low light intensities and generally return to the deeper, darker areas of a lake during daylight hours. Walleye would therefore be exposed to the same high ambient mercury levels as pike, but on less frequent basis. Despite this lower exposure time to high mercury, walleye nevertheless appear to have similar or even higher mercury concentrations than pike (Fig. 35). The higher mercury concentration in walleye may be due to, as noted by Mathers and Johansen (1985), the greater activity level, higher food consumption rate and lower food conversion efficiency of walleye as oppposed to pike. These factors would predispose walleye to a higher degree of uptake of mercury from both the water and food sources throughout the whole lake. The walleye, therefore, is perhaps a more effective integrator of mercury contamination in the freshwater environment. As such, walleye, more than other fish species, may be expected to more accurately reflect the increase in mercury levels in lakes caused by flooding.

As inferred in the foregoing discussion, a variety of factors ranging from variations in local conditions to differences in fish species characteristics may influence the empirical relationship between fish mercury concentrations and the degree of flooding on a lake. The number and complexity of factors which may influence this relationship are further illustrated in a review of the causes of increased mercury releases in reservoirs by Jones _et al_. (1986). We also found that sample variability, which may result from an inadequate representation of fish sizes or numbers or of fish from all areas of a lake, may influence the relationship. It is therefore not surprising that we could not derive a significant quantifiable relationship, which would serve to predict, with reasonable confidence the elevation of fish mercury concentrations in impounded waters based solely on the extent of flooding. This agrees with the finding of Jones _et al_. (1986). They attempted to correlate relative flooded area[1], along with a number of other factors, to post-impoundment mercury levels in fish, but found that its incremental contribution to the overall r^2 values for whitefish, walleye and northern pike was quite small. We also compared our fish mercury data to relative flooded areas, but found that it did not significantly improve the correlations. We did, however, find significant correlations when both parameters were ranked (Fig. 36). This would indicate that there is an underlying, direct cause and effect relationship between the amount of area flooded, the amount of mercury released or made available for bioaccumulation and consequently the degree to which mercury levels are elevated in most fish. This contention is partly corroborated by results of Abernathy _et al_. (1985), who found that the concentrations of total mercury in water in a newly impounded reservoir were closely related to the increases in water surfaces area.

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[1] Relative flooded area was derived by dividing the lake surface area increase following impoundment by the post-impoundment surface area. We determined the extent of flooding by dividing the post-impoundment surface area increase by the pre-impoundment surface area.

It should be noted that the best rank correlation between extent of flooding and mercury concentrations was obtained for walleye ($r^2 = 0.648$). This coincides with the results of Jones _et al_. (1986), who examined the relationship of a number of reservoir site parameters, including relative flooded area, with post-impoundment mercury levels in whitefish, walleye and pike and found the best overall correlation was achieved for walleye ($r^2 = 0.715$). They also found that most of the correlation in the case of walleye mercury levels was accounted for by the mean post-impoundment depths of the reservoirs; i.e., as the mean depth of the reservoir, and consequently the extent of flooded shoreline increased, the fish mercury concentrations increased proportionately. These observations support our contention that walleye are more effective integrators of lake-wide levels in ambient mercury.

Although we could not derive a precise predictive equation, the empirical results nevertheless allowed for a crude estimate of the amount of fish mercury levels would be elevated given varying degrees of flooding. The relationships of the length adjusted mercury concentrations in whitefish, walleye and pike to extent of flooding are shown in Fig. 40. The numbers in the symbols coincide with the lake names in the order listed in Table 10. Note that two lakes, Notigi (7) and Stephens (19), are deleted from Fig. 40, since both were flooded by more than 200 per cent. The dashed line curves, which were fitted by inspection, represent the average or most probable relationship between flooding and mercury levels for each species. The solid line curves generally represent the range in average fish mercury levels that might be anticipated at a given level of flooding. A common envelope is drawn for both walleye and pike because of the overlap in mean mercury levels in these species (Table 4). The upper and lower limits of this "confidence" envelope have been arbitrarily drawn. The upper limit, which levels off around 2.0 $\mu g.g^{-1}$ mercury, however, is supported by our results. Mean mercury levels in walleye and pike samples taken in the present study approached, but did not exceed the 2.0 $\mu g.g^{-1}$ level. The "confidence" limits about the median line for whitefish were also arbitrarily drawn. Mercury concentrations in one or more of the three considered species from a number of lakes fall outside of the "confidence" envelopes. Results for Mynarski (3), Wapisu (8) and Mystery (15) lakes are consistent in that the mercury concentrations in all three

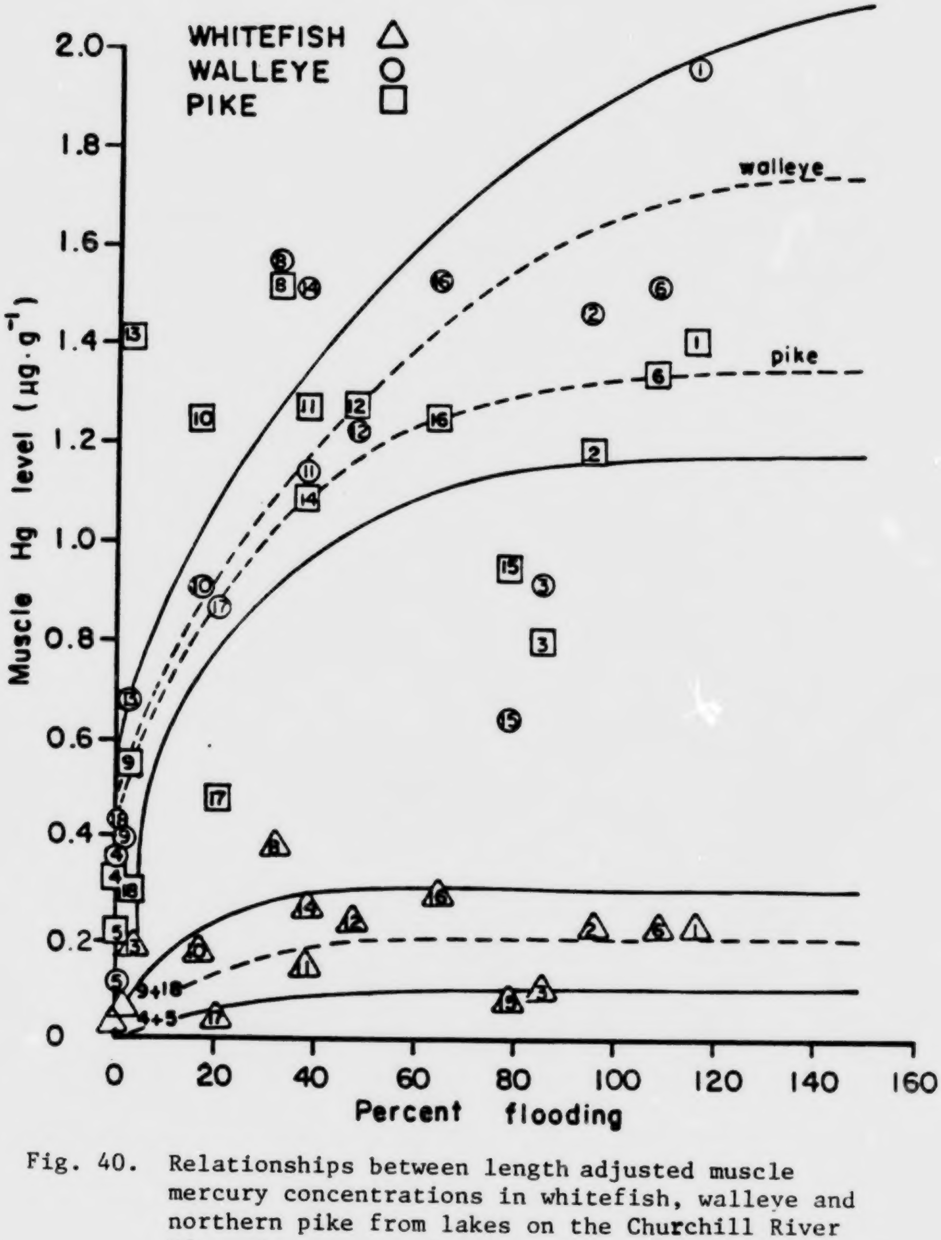

Fig. 40. Relationships between length adjusted muscle mercury concentrations in whitefish, walleye and northern pike from lakes on the Churchill River Diversion route and Nelson River and the extent of flooding on each lake. Lake numbers coinciding with lake names in the order presented in Table 10 are given in the symbols. Points for Notigi (7) and Stephens (19) are not shown. See text for explanation of eye-fitted lines.

species from these lakes are obviously unusual from those for other lakes. A possible reason for the high mercury levels in Wapisu Lake has already been offered. The low mercury levels in Mynarski Lake relative to the extent of flooding could be the result of an influx of low mercury fish from the eastern portion of the lake which was not flooded. We cannot account for the unusual mercury levels in Mystery Lake. In addition, mercury concentrations in pike for Osik (10), Wuskwatim (13) and Sipiwesk (17) lakes and in Birch Tree Lake walleye (14) were outside the "confidence" envelope. Except for Wuskwatim Lake pike, these unusual results are also difficult to explain. Mercury concentrations in Sipiwesk Lake whitefish were also lower than might be expected.

The cluster of observations along the ordinate in Fig.40 indicate that the background mercury levels are between 0.3 and 0.4 $\mu g.g^{-1}$ for both walleye and pike and less than 0.1 $\mu g.g^{-1}$ for whitefish. This agrees closely with the range in mean mercury levels found in fish samples from unimpacted lakes within the general study area (Fig. 37). When lakes are flooded even a small amount (less than 5%), mercury concentrations in walleye and pike may be expected to increase as much as 50 per cent. In lakes subjected to minor flooding, higher mercury levels may be realized in pike than in walleye , because of the pike's preference for inshore flooded habitats. The elevation in whitefish mercury levels in situations of minor flooding is negligible, if it occurs at all. This is a direct result of the whitefish's preference for mid-lake areas where mercury methylation rates are not generally influenced by flooding. When lake surface areas are increased between 10 and 20 per cent, mercury concentrations in walleye , and perhaps pike, appear to increase by over 100 per cent over background levels. Increases of a similar magnitude are also apparent in whitefish mercury levels with the same degree of flooding. When flooding reaches 50 per cent, there is approximately a three-fold increase in mercury concentrations in both walleye and northern pike (Fig. 40). After this stage, the observed results suggest there is a divergence in the mercury content of the two species. Mercury concentrations in pike appear to level off at about 1.20 $\mu g.g^{-1}$, and may remain there irrespective of the degree of flooding. In Notigi Lake where flooding exceeded 400 per cent, the mercury level in pike was 1.19 $\mu g.g^{-1}$. Mercury levels in walleye may level off somewhere between 1.40 and 1.60 $\mu g.g^{-1}$; Notigi Lake walleye had a mercury concentration of 1.39 $\mu g.g^{-1}$. Mercury levels in pike and walleye from Stephens Lake, where flooding reached 237 per cent, were lower (0.81 and 0.87 $\mu g.g^{-1}$, respectively) than the above equilibriums. However, because of geological and water quality differences, Stephens Lake results may not be strictly comparable to those for the Rat-Burntwood lakes. The mercury content in whitefish appeared to level out at approximately 0.2 $\mu g.g^{-1}$.

As indicated by the wide range in mercury concentrations observed in Fig. 40, the association between flooding and fish mercury increases, particularly in walleye and pike, appears to break down as the degree of flooding approaches and exceeds 100 per cent. It seems logical that after the point is reached where there is almost as much flooded land as former lake bed, mercury levels in walleye and pike would not be closely correlated to the degree of flooding. Extensive flooding usually occurs when there is little relief in the land surrounding the lake or river. Water over such newly flooded areas may be too shallow to provide suitable habitat for fish. Because of the shallow depths, the area would freeze out in winter and be too warm in summer, at least for coolwater species such as pike and walleye . In such cases, fish would tend to occupy pre-existing lake areas where the release of mercury may not be as pronounced. Where flooded areas may still have enough depth to support fish, high erosion and suspension of inorganic material through wind action may reduce methylation rates (Jones _et al_. 1986) and consequently result in a reduced elevation of mercury levels in fish. Mercury releases also may not be as great in flooded marshy areas surrounding lakes (Ramsey and Ramlal 1986). It therefore seems reasonable to assume that the level of mercury contamination in fish will not progressively increase with increased flooding. At some point an equilibrium in fish mercury levels will be reached. The level of that equilibrium will depend on the intrinsic level of environmental mercury available for mobilization into the water column and aquatic food chain, the suite of circumstances or conditions peculiar to the area which may influence the methylation and uptake of mercury, and, of course, the fish species.

Our results on the association of fish mercury levels with flooded areas appear to be similar to those of Jones et al. (1986), who examined the question from a slightly different perspective. They related fish mercury levels not only to relative flooded area, but to the index "flooded area concentration"; i.e., the flooded area divided by the reservoir volume. This index does not increase appreciably where the lake basin is quite deep; in these cases even a major increase in reservoir volume may cause only minor shoreline flooding. Where a lake basin is more saucer shaped, an extensive area may be flooded with only a small change in volume; the index in this event will be quite large. Although Jones et al. (1986) concluded there was no positive association between flooded area concentration and mercury levels in pike and walleye, a relationship may indeed exist. The data referred to by Jones et al. (1986) are presented in Fig. 41; it should be noted that the data represent reservoirs from a number of geological and climatic regions of Canada. If the reservoirs with the three highest mercury concentrations are omitted, the association of mercury in both pike and walleye with flooded area concentration may be described by a parabolic curve. If this association is genuine, it suggests that there is a very rapid increase in fish mercury levels with minor amounts of flooding and that with extensive inundation, the increase in fish mercury reaches some equilibrium or even shows slight decline from peak levels. Because of the wide variety of reservoir types considered by

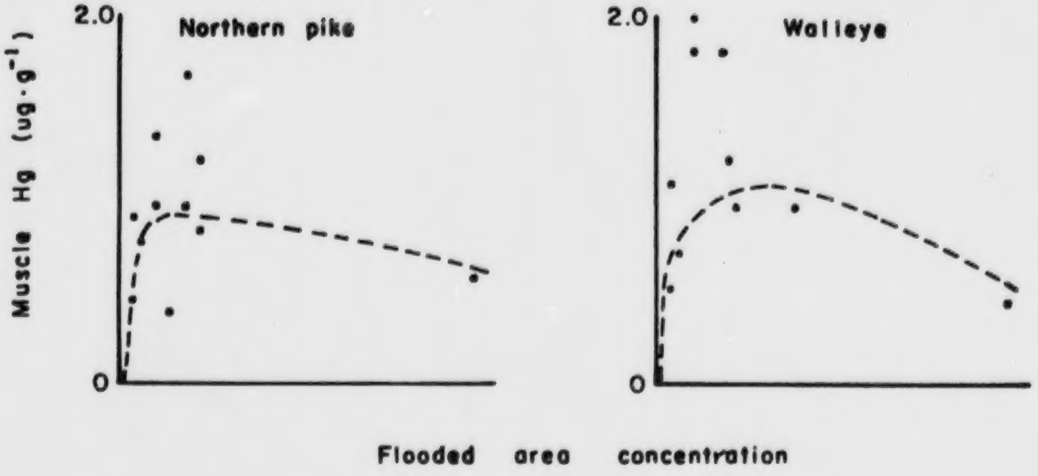

Fig. 41. The association between mercury concentrations in northern pike and walleye and flooded area concentrations for a number of reservoirs in Canada (after Jones *et al.* 1986). Flooded area concentration was derived by dividing the flooded area by the reservoir volume. Parabolas were eye-fitted to the data points.

Jones et al. (1986) we may be interpreting too much from their data. However, the above mentioned similarities seem to us to be more than coincidental. We therefore conclude that the extent of flooding is closely associated to and may, to a certain point, be used to predict the extent to which mercury levels are elevated in fish.

Plans for future hydroelectric developments in northern Manitoba call for several dams along the Burntwood River; dams have been proposed at the outlet of Wuskwatim Lake, at Manasan Falls near Thompson, and at First Rapids near the confluence of the Burntwood and Odei rivers. A small commercial fishery presently exists in Wuskwatim Lake, where fish mercury levels are low enough to allow the fish to sold on export markets. Sport fisheries would likely develop on the Manasan and First Rapids reservoirs as both areas are road accessible from Thompson. Because of these sport and commercial fishing interests, it is important that an attempt be made to predict the extent that fish mercury levels would increase in these areas. This prediction could be made if the degree of flooding on each of the reservoirs was known.

In addition to being able to predict the effect of flooding on fish mercury levels, it is also important to determine how long the elevated mercury levels will prevail. Investigations such as those by Abernathy and Cumbie (1977) and Meister et al. (1979) have concluded that elevated mercury levels in fish in newly impounded reservoirs may be a transitory phenomenon. This may in fact be the case in southern reservoirs. Pike in the Shellmouth reservoir on the Assiniboine River in southwestern Manitoba exhibited an increase in mercury levels (Fig. 42) within a year or two of impoundment (Derksen 1973). Unfortunately, these 1971-1972 levels cannot be compared to current pike mercury concentrations, since pike in the reservoir are very sparse at the present time and adequate samples are difficult to obtain. However, comparison of mercury levels in white suckers taken from Shellmouth reservoir in 1971-1972 to those taken during 1982-1984 by Beck (1984 and unpublished data) suggests that a decline has occurred over the past 10 years (Fig. 43). Declines in fish mercury levels from the present study area may not occur as quickly. Fish mercury levels in reservoirs in northern boreal forest regions may remain elevated for much longer times. Lodenius _et al_. (1983) observed elevated mercury concentrations in fish as much as 14 years after impoundment in two reservoirs in Finland. Other studies under the present Agreement (eg., Hecky _et al_. 1986, Ramsey and Ramlal 1986) have examined this question with reference to the Churchill River Diversion and other northern Manitoba reservoirs and have concluded that the problem will persist for several decades or more. These studies have found continuing high mercury methylation rates in Reindeer and Stephens lakes which were impounded in 1942 and 1971, respectively. This conclusion has been corroborated in part by the observation of little or no change in mercury levels in walleye and northern pike from the Rat River lakes which have been inundated for 10 to 12 years and from Sipiwesk Lake which has been flooded for more than 20 years.

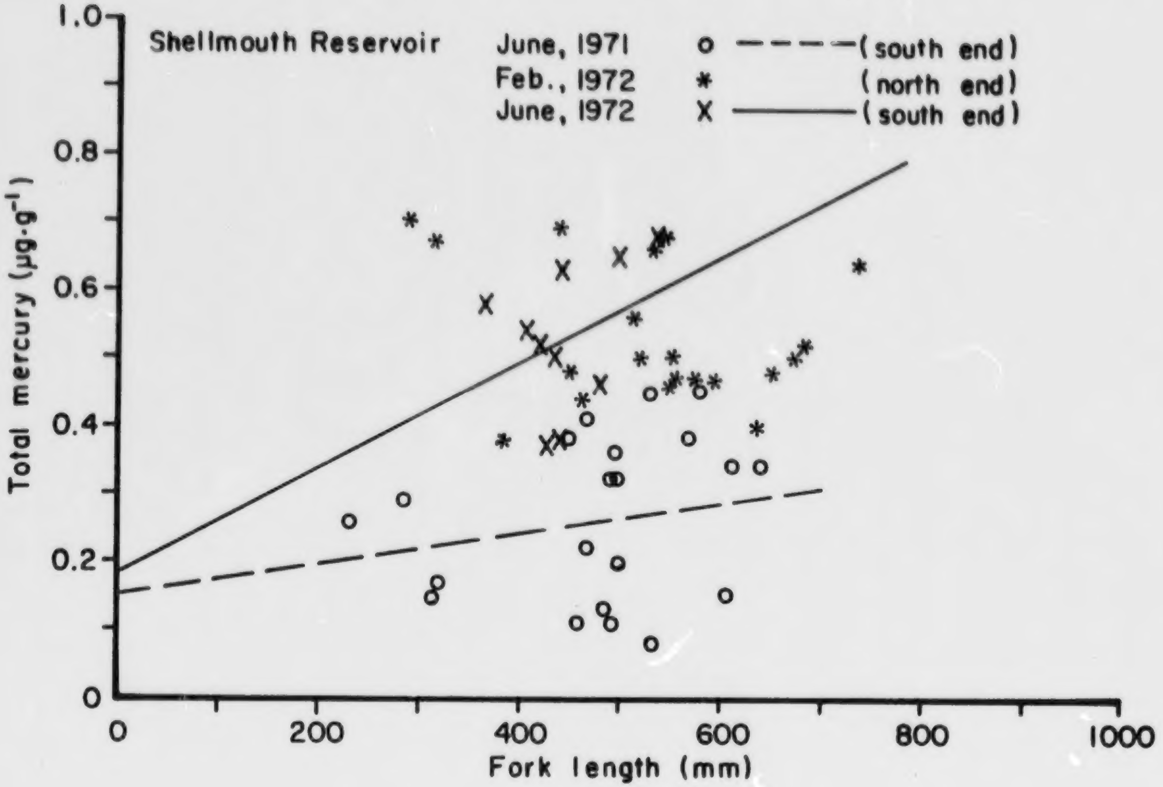

Fig. 42. Muscle mercury concentrations in northern pike taken from the Shellmouth Reservoir in southwestern Manitoba during 1971 and 1972. Lines fitted by least squares regression.

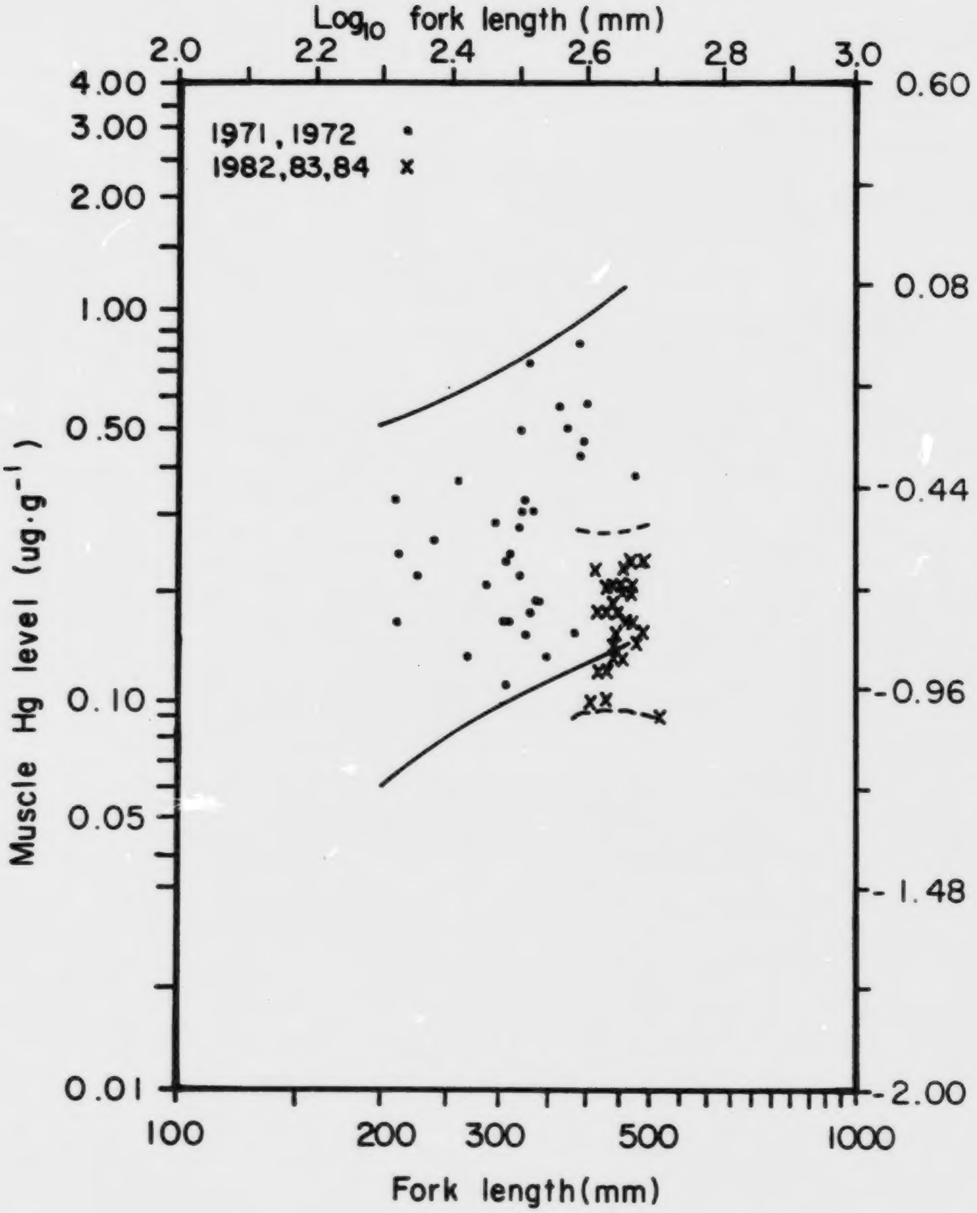

Fig. 43. Muscle mercury concentrations in white suckers taken from Shellmouth Reservoir in southwestern Manitoba during the periods 1971 - 1972 and 1982 - 1984. The 95% confidence intervals about the sample data are shown. The 1982 - 1984 data were provided by A. Beck, Environmental Management Division, Manitoba Department of Environment and Workplace Health and Saftey.

CONCLUSIONS AND RECOMMENDATIONS

The highest muscle mercury concentrations observed in fishes from lakes associated with the Chruchill River Diversion and Nelson River were recorded in the piscivorous species; i.e. northern pike, walleye and sauger. Mercury concentrations in some of the more highly contaminated individuals of these species exceeded 2.5 $\mu g{\cdot}g^{-1}$. In terms of numbers, pike, walleye and sauger constituted nearly 30 per cent of fish catches from the study area. Although occasional individuals of the 10 other large fish species which were tested contained muscle mercury levels between 0.5 and 1.1 $\mu g{\cdot}g^{-1}$, sample means for these species were typically less than the 0.5 $\mu g{\cdot}g^{-1}$ tolerance level.

Mercury concentrations in the axial muscles of whitefish and walleye were best correlated with fish age. This relationship was most likely attributable to the fact that both these species, particularly from moderately or lightly exploited populations in northern Manitoba, may attain relatively advanced ages without any significant growth in length. The exceptionally high mercury concentrations noted in both whitefish and walleye were usually found among these older, slow growing individuals. Fork length provided the best correlation with mercury content in pike. This is because of the great overlap in length that may occur in pike of different age groups.

The relationship between fish size or age and muscle mercury concentrations is primarily exponential in nature. This is due to the much lower concentrations found in the young fish. The lower mercury concentrations in the young fish may be a consequence of several factors such as differences in diet and metabolic rates, as well as exposure times between young and adult fish.

Covariance analysis of the fork length-mercury relationships in whitefish, walleye and northern pike more frequently produced significant differences between regression line elevations than in regression line slopes. From these results, it was concluded that the rates of mercury accumulation in whitefish, walleye and pike, although species dependent, were largely constant from year to year and from lake to lake within the study area. Variations in the elevations of the regression lines indicate areal or temporal differences in the amount of environmental mercury available for uptake.

Mercury levels in whitefish, walleye and northern pike were highest in the Notigi reservoir area. Lakes in the reservoir which were influenced by diversionary flows experienced the greatest degree of flooding of any of the lakes in the study area. The increase in surface area of lakes in the Notigi forebay varied between 85 and 435 per cent. Except for three lakes, flooding on all other lakes in the study area was less than 50 per cent. The exceptions were Mystery, Apussigamasi and Stephens lakes, where flooding ranged between 64 and 237 per cent. The flooding of lakes directly or indirectly influenced by the Churchill River Diversion has caused mercury contamination of walleye and pike from these lakes to increase as much as five times over the apparent background mercury levels in these species. Mercury levels in whitefish from impacted lakes approximately doubled as a result of flooding. Background mercury levels in pike and walleye from unimpacted lakes in or adjacent to the study area varied between 0.3 and 0.4 $\mu g \cdot g^{-1}$, while in whitefish they were less than 0.1 $\mu g \cdot g^{-1}$.

The increases in mercury contamination levels in both pike and walleye were directly related to the amount of land flooded relative to non-flooded water surface area. This relationship was most significant in the case of walleye. The walleye, because of its metabolic requirements and behavioural characteristics, is a more effective integrator of environmental mercury than is either pike or whitefish. Since pike prefer flooded inshore areas where mercury methylation rates are high, they tend in some cases to exhibit excessively high contamination levels. The opposite pertains to whitefish which tend to occupy offshore areas where the bacterial synthesis of methylmercury is generally low. The relationships between fish mercury levels and degree of flooding was most evident in areas where flooding did not exceed 100 per cent. After this point, mercury residues in both walleye and pike did not continue to elevate with increased flooding.

The level of mercury contamination in fishes from flooded lakes along the Churchill River Diversion does not appear to have declined significantly in the 10 years following impoundment and diversion. Fish mercury information examined in the present study, as well as available evidence from other northern Manitoba reservoirs, suggests that such declines may not materialize for at least several decades.

Based on the findings of the present study the following recommendations are made:

1. Residents along the Churchill River Diversion should be cautioned against the continuous and prolonged consumption of northern pike,

walleye and sauger from flooded lakes on the Rat - Burntwood system. They should particularly be advised to avoid consumption of the larger pike and walleye and all sizes of sauger , which are the most highly contaminated.

2. If facilities are developed on the Churchill River Diversion to exploit sport fishing potential, under no circumstances should fishing guides employed in such operations be allowed to consume shore lunches of fish.
3. Since the principle commercial species in order of relative abundance in the Churchill River Diversion lakes are whitefish, walleye and northern pike and since the latter two are too heavily contaminated to be marketable, any commercial fishing permitted in the area should be restricted to whitefish. However, further evaluations are required to determine whether or not such a fishery would be economically viable or could be selective for whitefish only.
4. An adequate degree of monitoring mercury levels in fish, particularly walleye, from lakes on the Churchill River Diversion should be maintained to document any trends in the level of mercury contamination and to provide further information for impact assessment of any future reservoir developments on the Rat - Burntwood system.

ACKNOWLEDGEMENTS

The authors would like to extend their thanks to the many people who were involved in the collection of fish samples during this study. The assistance of North Eastern Region Fisheries Branch staff is particularly appreciated. We would like to thank Drew Bodaly for providing references on mercury contamination in reservoirs. Walt Lysack provided advice on statistical analysis.

A special thanks is extended to Lorna McKerness for her assistance and encouragement throughout the course of this study.

The assistance of Coreene Caverly, Merle McTaggart, Cindy Little and Barb Bean in typing this report is greatly appreciated.

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APPENDIX I

List of common and scientific names of fish species captured by gillnetting in the present study:

COMMON NAME	SCIENTIFIC NAME
Lake whitefish	*Coregonus clupeaformis*
Walleye	*Stizostedion vitreum*
Sauger	*Stizostedion canadense*
Northern pike	*Esox lucius*
White sucker	*Catostomus commersoni*
Longnose sucker	*Catostomus catostomus*
Shorthead redhorse sucker	*Moxostoma macrolepidotum*
Yellow perch	*Perca flavescens*
Burbot	*Lota lota*
Freshwater drum	*Aplodinotus grunniens*
Mooneye	*Hiodon tergisus*
Goldeye	*Hiodon alosoides*
Lake chub	*Couesius plumbeus*
Trout-perch	*Percopsis omiscomaycus*
Cisco	*Coregonus artedii*

Appendix II : List of lakes which were not flooded by the Churchill River Diversion and for which fish muscle mercury data were available. See Fig. 2 for locations.

Lake		Drainage Basin
Name	No.	
Ulhman	1	1
Waskaiowaka	2	2
Costello	3	3
Suwannee	4	3
East Mynarski	5	3
Leftrock	6	3
Harding	7	4
Moak	8	4
Paint	9	5
Partridge Crop	10	5
Natawahunan	11	5
Brannigan	12	5
Pikwitonei	13	5
Cross	14	6
Walker	15	6
White Rabbit	16	6
Landing	17	7
Cauchon	18	7
Bearhead	19	7
Goulet	20	7
War	21	8
Butnau	22	8
Assean	23	8
Crying	24	8
Limestone	25	9

Appendix III. The following Tables present the mean fork length, the minimum, maximum and mean mercury concentrations of each sample, and the length adjusted mercury concentrations for lake whitefish, walleye and northern pike samples collected from lakes along the Churchill River Diversion, and Nelson River, 1972 - 1985. Adjusted mercury concentrations are estimated by regression analysis for 350 mm whitefish, 400 mm walleye and 550 mm northern pike.

Karsakuwigamak Lake.

Year	No. of fish	$\overline{FL}$ (mm)	Min. Hg ($\mu g \cdot g^{-1}$)	Max. Hg ($\mu g \cdot g^{-1}$)	$\overline{Hg}$ ($\mu g \cdot g^{-1}$)	Adjusted Hg level ($\mu g \cdot g^{-1}$)	% > 0.54 $\mu g \cdot g^{-1}$	% > 1.0 $\mu g \cdot g^{-1}$
					Whitefish			
1984	53	358	0.13	0.62	0.23	0.22	1.89	0.0
					Walleye			
1984	51	371	0.54	3.52	1.84	1.96	98.0	73.0
					Northern Pike			
1984	52	671	0.39	4.73	1.99	1.41	98.0	90.0

Pemichigamau Lake.

Year	No. of fish	$\overline{FL}$ (mm)	Min. Hg ($\mu g \cdot g^{-1}$)	Max. Hg ($\mu g \cdot g^{-1}$)	$\overline{Hg}$ ($\mu g \cdot g^{-1}$)	Adjusted Hg level ($\mu g \cdot g^{-1}$)	% > 0.54 $\mu g \cdot g^{-1}$	% > 1.0 $\mu g \cdot g^{-1}$
					Whitefish			
1985	36	338	0.12	0.33	0.22	0.22	0.0	0.0
					Walleye			
1985	55	382	0.28	2.80	1.45	1.47	90.9	72.7
					Northern Pike			
1985	56	674	0.23	2.95	1.63	1.19	94.6	85.7

West Mynarski Lake.

Year	No. of fish	$\overline{FL}$ (mm)	Min. Hg ($\mu g \cdot g^{-1}$)	Max. Hg ($\mu g \cdot g^{-1}$)	$\overline{Hg}$ ($\mu g \cdot g^{-1}$)	Adjusted Hg level ($\mu g \cdot g^{-1}$)	% > 0.54 $\mu g \cdot g^{-1}$	% > 1.0 $\mu g \cdot g^{-1}$
					Whitefish			
1984	52	358	0.03	0.66	0.14	0.10	1.92	0
					Walleye			
1982	33	461	0.24	2.99	1.23	1.01	85	42
1983	14	438	0.32	1.21	0.63	0.41	57	7
1984	49	363	0.15	2.45	0.95	0.92	51	47
					Northern Pike			
1982	12	616	0.30	0.82	0.54	0.49	42	0
1983	5	745	0.22	0.77	0.48	2.06	20	0
1984	51	625	0.12	2.48	1.20	0.80	84	57

Costello Lake.

Year	No. of fish	$\overline{FL}$ (mm)	Min. Hg ($\mu g \cdot g^{-1}$)	Max. Hg ($\mu g \cdot g^{-1}$)	$\overline{Hg}$ ($\mu g \cdot g^{-1}$)	Adjusted Hg level ($\mu g \cdot g^{-1}$)	% > 0.54 $\mu g \cdot g^{-1}$	% > 1.0 $\mu g \cdot g^{-1}$
					Whitefish			
1985	20	430	0.03	0.13	0.07	0.04	0	0
					Walleye			
1979	6	431	0.21	1.51	0.51		17	17
1985	53	460	0.13	0.87	0.52	0.37	42	0
					Northern Pike			
1985	54	476	0.06	0.58	0.26	0.33	4	0

Suwannee Lake.

Year	No. of fish	$\overline{FL}$ (mm)	Min. Hg ($\mu g \cdot g^{-1}$)	Max. Hg ($\mu g \cdot g^{-1}$)	$\overline{Hg}$ ($\mu g \cdot g^{-1}$)	Adjusted Hg level ($\mu g \cdot g^{-1}$)	% > 0.54 $\mu g . g^{-1}$	% > 1.0 $\mu g . g^{-1}$
					Whitefish			
1985	56	398	0.01	0.09	0.04	0.03	0	0
					Walleye			
1979	6	525	0.64	1.22	0.95		100	50
1985	34	416	0.03	0.36	0.14	0.11	0	0
					Northern Pike			
1979	7	512	0.13	0.23	0.17		0	0
1985	56	495	0.02	0.70	0.19	0.21	1.79	0

Rat Lake whitefish.

Year	No. of fish	$\overline{\mathrm{FL}}$ (mm)	Min. Hg ($\mu g.g^{-1}$)	Max. Hg ($\mu g.g^{-1}$)	$\overline{\mathrm{Hg}}$ ($\mu g.g^{-1}$)	Adjusted Hg level ($\mu g.g^{-1}$)	% > 0.54 $\mu g.g^{-1}$	% > 1.0 $\mu g.g^{-1}$
1978	5	439	0.26	0.59	0.40	0.62	20.0	0.0
1980	24	422	0.15	0.61	0.32	0.23	4.0	0.0
1983	23	415	0.13	0.70	0.29	0.18	9.0	0.0
1984	50	333	0.07	1.00	0.26	0.25	2.0	0.0
1985	53	354	0.08	0.86	0.24	0.21	5.7	0.0

Rat Lake walleye.

Year	No. of fish	$\overline{\mathrm{FL}}$ (mm)	Min. Hg ($\mu g \cdot g^{-1}$)	Max. Hg ($\mu g \cdot g^{-1}$)	$\overline{\mathrm{Hg}}$ ($\mu g \cdot g^{-1}$)	Adjusted Hg level ($\mu g \cdot g^{-1}$)	% > 0.54 $\mu g \cdot g^{-1}$	% > 1.0 $\mu g \cdot g^{-1}$
1978	26	449	2.17	3.51	2.56	2.36	100	100
1979	25	450	1.68	3.29	2.32	2.08	100	100
1980	22	408	0.41	3.37	1.15	1.00	95	45
1983	50	377	0.34	4.10	1.79	1.85	92	72
1984	65	327	0.27	3.89	1.55	1.92	83	62
1985	54	388	0.41	3.29	1.55	1.52	91	65

Rat Lake northern pike.

Year	No. of fish	$\overline{FL}$ (mm)	Min. Hg ($\mu g \cdot g^{-1}$)	Max. Hg ($\mu g \cdot g^{-1}$)	$\overline{Hg}$ ($\mu g \cdot g^{-1}$)	Adjusted Hg level ($\mu g \cdot g^{-1}$)	% > 0.54 $\mu g \cdot g^{-1}$	% > 1.0 $\mu g \cdot g^{-1}$
1978	24	698	1.47	2.49	2.05	1.92	100	100
1980	1	628	2.37	2.37	2.37	-	100	100
1983	22	599	0.30	2.45	1.54	1.33	95	91
1984	52	601	0.38	3.75	1.86	1.52	96	83
1985	58	608	0.49	3.21	1.60	1.34	98	79

Notigi Lake whitefish.

Year	No. of fish	$\overline{FL}$ (mm)	Min. Hg ($\mu g \cdot g^{-1}$)	Max. Hg ($\mu g \cdot g^{-1}$)	$\overline{Hg}$ ($\mu g \cdot g^{-1}$)	Adjusted Hg level ($\mu g \cdot g^{-1}$)	% > 0.54 $\mu g \cdot g^{-1}$	% > 1.0 $\mu g \cdot g^{-1}$
1980	6	421	0.07	0.25	0.12	0.39	0	0
1981	38	309	0.12	0.71	0.23	0.25	3	0
1983	50	350	0.06	0.48	0.20	0.19	0	0
1985	50	314	0.15	0.70	0.26	0.28	2	0

Notigi Lake walleye.

Year	No. of fish	$\overline{FL}$ (mm)	Min. Hg ($\mu g.g^{-1}$)	Max. Hg ($\mu g.g^{-1}$)	$\overline{Hg}$ ($\mu g.g^{-1}$)	Adjusted Hg level ($\mu g.g^{-1}$)	% > 0.54 $\mu g.g^{-1}$	% > 1.0 $\mu g.g^{-1}$
1978	19	422	0.19	2.91	1.41	0.80	53	53
1980	4	453	2.11	3.47	2.90	2.05	100	100
1981	29	416	0.95	2.55	1.88	1.79	100	97
1982	6	503	0.98	1.63	1.23	0.79	100	83
1983	26	370	0.40	2.95	1.35	1.44	96	58
1985	25	385	0.34	2.58	1.20	1.19	88	48

Notigi Lake northern pike.

Year	No. of fish	$\overline{FL}$ (mm)	Min. Hg (μg.g^{-1})	Max. Hg (μg.g^{-1})	$\overline{Hg}$ (μg.g^{-1})	Adjusted Hg level (μg.g^{-1})	% > 0.54 g.g^{-1}	% > 1.0 g.g^{-1}
1980	5	788	1.62	2.29	1.95	2.58	100	100
1981	50	581	0.24	2.82	1.70	1.48	98	98
1982	10	739	1.32	2.38	1.85	2.24	100	100
1983	44	515	0.29	2.23	1.18	1.23	77	61
1985	55	657	0.56	2.87	1.72	1.39	100	96

Wapisu Lake.

Year	No. of fish	$\overline{FL}$ (mm)	Min. Hg ($\mu g \cdot g^{-1}$)	Max. Hg ($\mu g \cdot g^{-1}$)	$\overline{Hg}$ ($\mu g \cdot g^{-1}$)	Adjusted Hg level ($\mu g \cdot g^{-1}$)	% > 0.54 $\mu g.g^{-1}$	% > 1.0 $\mu g.g^{-1}$
					Whitefish			
1983	41	360	0.09	0.82	0.31	0.27	7	0
1985	39	328	0.16	1.05	0.39	0.38	21	3
					Walleye			
1977	91	411	0.14	3.03	1.17	1.00	92	57
1983	50	276	0.33	2.46	1.05	1.42	78	56
1985	64	366	0.29	2.34	1.42	1.56	97	84
					Northern Pike			
1977	38	672	0.32	2.25	1.08	0.61	71	47
1983	53	645	0.57	3.42	1.97	1.68	100	96
1985	41	598	0.40	3.38	1.76	1.53	95	88

Macheewin Lake.

Year	No. of fish	$\overline{FL}$ (mm)	Min. Hg ($\mu g \cdot g^{-1}$)	Max. Hg ($\mu g \cdot g^{-1}$)	$\overline{Hg}$ ($\mu g \cdot g^{-1}$)	Adjusted Hg level ($\mu g \cdot g^{-1}$)	% > 0.54 $\mu g . g^{-1}$	% > 1.0 $\mu g . g^{-1}$
					Whitefish			
1984	50	354	0.02	0.25	0.08	0.06	0	0
					Walleye			
1981	6	478	0.33	0.89	0.66	1.60	67	0
1982	6	475	0.37	0.96	0.76	1.09	83	0
1984	44	352	0.08	0.86	0.36	0.40	16	0
					Northern Pike			
1981	5	661	0.29	0.45	0.35	-	0	0
1982	10	683	0.41	1.78	0.87	0.31	70	30
1984	57	522	0.08	1.50	0.56	0.55	46	5

Osik Lake.

Year	No. of fish	$\overline{FL}$ (mm)	Min. Hg ($\mu g \cdot g^{-1}$)	Max. Hg ($\mu g \cdot g^{-1}$)	$\overline{Hg}$ ($\mu g \cdot g^{-1}$)	Adjusted Hg level ($\mu g \cdot g^{-1}$)	% > 0.54 $\mu g \cdot g^{-1}$	% > 1.0 $\mu g \cdot g^{-1}$
					Whitefish			
1984	58	370	0.07	0.54	0.19	0.17	0	0
					Walleye			
1982	6	438	1.00	1.23	1.12		100	83
1983	18	405	0.05	1.05	0.56	0.79	61	22
1985	51	343	0.20	1.68	0.79	0.91	59	33
					Northern Pike			
1982	5	724	1.12	1.36	1.24	1.17	100	100
1983	5	713	0.41	2.21	1.67	1.34	80	80
1984	60	498	0.22	2.76	1.15	1.25	75	57

Footprint Lake whitefish.

Year	No. of fish	$\overline{FL}$ (mm)	Min. Hg ($\mu g \cdot g^{-1}$)	Max. Hg ($\mu g \cdot g^{-1}$)	$\overline{Hg}$ ($\mu g \cdot g^{-1}$)	Adjusted Hg level ($\mu g \cdot g^{-1}$)	% > 0.54 $\mu g \cdot g^{-1}$	% > 1.0 $\mu g \cdot g^{-1}$
1980	8	340	0.18	0.44	0.27	0.24	0	0
1981	22	316	0.09	0.26	0.17	0.18	0	0
1983	25	366	0.06	0.48	0.18	0.15	0	0
1984	52	359	0.03	0.31	0.11	0.09	0	0
1985	53	341	0.06	0.42	0.15	0.14	0	0

Footprint Lake walleye.

Year	No. of fish	$\overline{FL}$ (mm)	Min. Hg ($\mu g \cdot g^{-1}$)	Max. Hg ($\mu g \cdot g^{-1}$)	$\overline{Hg}$ ($\mu g \cdot g^{-1}$)	Adjusted Hg level ($\mu g \cdot g^{-1}$)	% > 0.54 $\mu g \cdot g^{-1}$	% > 1.0 $\mu g \cdot g^{-1}$
1978	20	382	0.70	2.61	0.98	1.01	100	20
1980	6	394	0.73	1.72	1.25	1.20	100	83
1981	30	376	0.71	1.76	1.10	1.21	100	67
1983	33	353	0.36	1.85	1.13	1.23	91	58
1984	54	379	0.31	1.87	1.13	1.17	81	72
1985	57	253	0.28	1.50	0.70	1.14	44	23

Footprint Lake northern pike.

Year	No. of fish	$\overline{FL}$ (mm)	Min. Hg ($\mu g \cdot g^{-1}$)	Max. Hg ($\mu g \cdot g^{-1}$)	$\overline{Hg}$ ($\mu g \cdot g^{-1}$)	Adjusted Hg level ($\mu g \cdot g^{-1}$)	% > 0.54 $\mu g \cdot g^{-1}$	% > 1.0 $\mu g \cdot g^{-1}$
1978	18	455	0.44	0.90	0.70	0.71	94	0
1980	4	526	1.06	1.53	1.20		100	100
1981	14	520	0.83	1.74	1.12	1.15	100	100
1983	41	477	0.45	2.40	1.20	1.35	93	59
1984	51	444	0.18	3.59	1.02	1.48	80	41
1985	48	431	0.29	1.73	0.85	1.27	81	35

Threepoint Lake whitefish.

Year	No. of fish	$\overline{\text{FL}}$ (mm)	Min. Hg ($\mu g \cdot g^{-1}$)	Max. Hg ($\mu g \cdot g^{-1}$)	$\overline{\text{Hg}}$ ($\mu g \cdot g^{-1}$)	Adjusted Hg level ($\mu g \cdot g^{-1}$)	% > 0.54 $\mu g \cdot g^{-1}$	% > 1.0 $\mu g \cdot g^{-1}$
1980	5	429	0.34	0.52	0.39		0	0
1981	16	333	0.11	0.41	0.23	0.23	0	0
1983	25	366	0.10	0.56	0.24	0.21	4	0
1984	51	361	0.07	1.07	0.25	0.21	2	2
1985	53	339	0.09	0.55	0.24	0.23	2	0

Threepoint Lake walleye.

Year	No. of fish	$\overline{\text{FL}}$ (mm)	Min. Hg ($\mu g \cdot g^{-1}$)	Max. Hg ($\mu g \cdot g^{-1}$)	$\overline{\text{Hg}}$ ($\mu g \cdot g^{-1}$)	Adjusted Hg level ($\mu g.g^{-1}$)	% > 0.54 $\mu g \cdot g^{-1}$	% > 1.0 $\mu g \cdot g^{-1}$
1980	5	403	1.16	1.81	1.49	1.47	100	100
1981	42	381	0.84	1.05	1.35	1.40	100	93
1983	50	336	0.32	2.70	1.36	1.65	86	68
1984	54	351	0.33	2.56	1.15	1.31	81	59
1985	59	337	0.29	1.99	1.01	1.22	78	49

Threepoint Lake northern pike.

Year	No. of fish	$\overline{FL}$ (mm)	Min. Hg ($\mu g \cdot g^{-1}$)	Max. Hg ($\mu g \cdot g^{-1}$)	$\overline{Hg}$ ($\mu g \cdot g^{-1}$)	Adjusted Hg level ($\mu g \cdot g^{-1}$)	% > 0.54 $\mu g.g^{-1}$	% > 1.0 $\mu g.g^{-1}$
1980	5	688	0.71	2.08	1.18		100	40
1981	28	517	0.44	2.27	1.33	1.39	96	79
1983	50	493	0.25	3.53	1.58	1.75	86	76
1984	51	561	0.36	3.18	1.57	1.44	86	82
1985	49	532	0.29	2.63	1.28	1.27	84	76

Wuskwatim Lake.

Year	No. of fish	$\overline{FL}$ (mm)	Min. Hg ($\mu g \cdot g^{-1}$)	Max. Hg ($\mu g \cdot g^{-1}$)	$\overline{Hg}$ ($\mu g \cdot g^{-1}$)	Adjusted Hg level ($\mu g \cdot g^{-1}$)	% > 0.54 $\mu g.g^{-1}$	% > 1.0 $\mu g.g^{-1}$
					Whitefish			
1981	28	398	0.18	0.78	0.33	0.25	7	0
1984	56	339	0.04	0.51	0.20	0.18	0	0
					Walleye			
1979	57	415	0.64	2.18	0.92	0.87	100	25
1980	19	413	0.78	1.41	1.00	0.94	100	53
1981	34	399	0.62	1.36	0.89	0.88	100	29
1983	18	433	0.38	1.21	0.82	0.65	89	17
1984	57	367	0.14	2.20	0.66	0.68	54	19

Wuskwatim Lake northern pike.

Year	No. of fish	$\overline{FL}$ (mm)	Min. Hg ($\mu g.g^{-1}$)	Max. Hg ($\mu g.g^{-1}$)	$\overline{Hg}$ ($\mu g.g^{-1}$)	Adjusted Hg level ($\mu g.g^{-1}$)	% > 0.54 $\mu g.g^{-1}$	% > 1.0 $\mu g.g^{-1}$
					Northern Pike			
1979	55	562	0.34	2.77	0.88	0.82	91	24
1980	7	605	0.79	1.20	1.03	0.97	100	57
1981	25	514	0.47	1.98	0.80	0.85	84	12
1982	5	682	0.81	2.12	1.35	0.64	100	60
1984	38	450	0.12	2.49	1.02	1.42	66	45

Birch Tree Lake.

Year	No. of fish	$\overline{FL}$ (mm)	Min. Hg ($\mu g \cdot g^{-1}$)	Max. Hg ($\mu g \cdot g^{-1}$)	$\overline{Hg}$ ($\mu g \cdot g^{-1}$)	Adjusted Hg level ($\mu g \cdot g^{-1}$)	% > 0.54 $\mu g \cdot g^{-1}$	% > 1.0 $\mu g \cdot g^{-1}$
					Whitefish			
1985	34	343	0.13	0.44	0.27	0.26	0	0
					Walleye			
1985	17	320	0.45	2.08	1.13	1.51	76	59
					Northern Pike			
1985	20	568	0.37	2.37	1.17	1.09	95	60

Mystery Lake.

Year	No. of fish	$\overline{FL}$ (mm)	Min. Hg (μg·g)	Max. Hg (μg·g)	$\overline{Hg}$ (μg·g)	Adjusted Hg level (μg·g)	% > 0.54 μg·g	% > 1.0 μg·g
					Whitefish			
1983	21	343	0.05	0.45	0.19	0.17	0	0
					Walleye			
1979	33	462	0.53	1.75	1.13	1.01	97	58
1982	6	439	0.80	1.71	1.13	0.75	100	67
1983	21	416	0.13	1.59	0.82	0.65	67	38
					Northern Pike			
1979	45	541	0.23	1.64	0.79	0.85	69	29
1982	30	723	0.31	1.83	1.14	0.86	87	60
1983	50	601	0.51	2.05	1.06	0.95	96	50

Apussigamasi Lake.

Year	No. of fish	$\overline{FL}$ (mm)	Min. Hg ($\mu g \cdot g^{-1}$)	Max. Hg ($\mu g \cdot g^{-1}$)	$\overline{Hg}$ ($\mu g \cdot g^{-1}$)	Adjusted Hg level ($\mu g \cdot g^{-1}$)	% > 0.54 $\mu g \cdot g^{-1}$	% > 1.0 $\mu g \cdot g^{-1}$
					Whitefish			
1984	33	369	0.14	0.79	0.32	0.28	9	0
					Walleye			
1982	6	504	1.84	2.90	2.37	2.04	100	100
1983	5	530	1.36	2.10	1.77		100	100
1984	53	287	0.28	2.54	1.04	1.53	72	43
					Northern Pike			
1982	10	794	0.39	2.49	1.17	0.67	90	60
1983	5	734	1.02	1.69	1.40		100	100
1984	53	478	0.15	2.82	1.12	1.25	78	52

Sipiwesk Lake whitefish.

Year	No. of fish	$\overline{FL}$ (mm)	Min. Hg ($\mu g \cdot g^{-1}$)	Max. Hg ($\mu g \cdot g^{-1}$)	$\overline{Hg}$ ($\mu g \cdot g^{-1}$)	Adjusted Hg level ($\mu g . g^{-1}$)	% > 0.54 $\mu g \cdot g^{-1}$	% > 1.0 $\mu g \cdot g^{-1}$
1977	2	473	0.07	0.20	0.13		0	0
1983	16	396	0.01	0.12	0.05	0.04	0	0

Sipiwesk Lake walleye.

Year	No. of fish	$\overline{FL}$ (mm)	Min. Hg ($\mu g \cdot g^{-1}$)	Max. Hg ($\mu g \cdot g^{-1}$)	$\overline{Hg}$ ($\mu g \cdot g^{-1}$)	Adjusted Hg level ($\mu g \cdot g^{-1}$)	% > 0.54 $g \cdot g^{-1}$	% > 1.0 $g \cdot g^{-1}$
1972	54	414	0.54	1.40	0.84		98	15
1973	54	424	0.35	1.04	0.74	0.72	87	2
1974	22	447	0.11	1.12	0.68	0.53	68	9
1976	65	447	0.45	1.75	0.95	0.70	95	42
1977	173	443	0.13	1.66	0.75	0.57	72	23
1978	47	399	0.16	1.39	0.84	0.78	83	28
1979	54	439	0.16	2.28	0.86	0.63	78	39
1980	45	485	0.21	1.39	0.89		87	40
1981	271	430	0.09	2.16	0.78	0.58	70	25
1982	21	466	0.41	1.70	0.80	0.59	71	24
1983	56	332	0.07	1.21	0.63	0.87	63	14

Sipiwesk Lake northern pike.

Year	No. of fish	$\overline{FL}$ (mm)	Min. Hg ($\mu g.g^{-1}$)	Max. Hg ($\mu g.g^{-1}$)	$\overline{Hg}$ ($\mu g.g^{-1}$)	Adjusted Hg level ($\mu g.g^{-1}$)	% > 0.54 $\mu g.g^{-1}$	% > 1.0 $\mu g.g^{-1}$
1972	155	563	0.10	1.25	0.64	0.59	67	4
1973	141	620	0.36	1.29	0.67	0.58	70	2
1974	19	646	0.25	1.63	0.74	0.55	79	5
1976	157	564	0.24	1.97	0.66	0.59	55	14
1977	42	715	0.29	1.78	0.93	0.61	88	40
1978	12	516	0.44	1.09	0.72	0.69	67	25
1979	30	701	0.40	2.18	0.96	0.56	87	37
1980	97	684	0.42	3.55	0.94	0.61	95	29
1981	280	651	0.13	2.48	0.87	0.60	79	30
1982	215	688	0.14	2.93	1.00	0.61	84	44
1983	43	547	0.07	1.60	0.55	0.48	49	9

Assean Lake.

Year	No. of fish	$\overline{FL}$ (mm)	Min. Hg (µg·g)	Max. Hg (µg·g)	$\overline{Hg}$ (µg·g)	Adjusted Hg level (µg·g)	% > 0.54 µg·g	% > 1.0 µg·g
					Whitefish			
1981	43	385	0.02	0.13	0.06	0.04	0	0
1985	25	409	0.01	0.14	0.06	0.03	0	0
					Walleye			
1981	52	391	0.15	0.75	0.32	0.30	8	0
1982	6	448	0.24	0.48	0.33	0.26	0	0
1985	27	379	0.09	0.43	0.22	0.22	0	0
					Northern Pike			
1981	34	610	0.11	0.90	0.30	0.23	6	0
1982	5	805	0.29	0.92	0.51	0.27	40	0
1985	18	577	0.12	0.54	0.24	0.20	0	0

Split Lake whitefish.

Year	No. of fish	$\overline{FL}$ (mm)	Min. Hg ($\mu g.g^{-1}$)	Max. Hg ($\mu g.g^{-1}$)	$\overline{Hg}$ ($\mu g.g^{-1}$)	Adjusted Hg level ($\mu g.g^{-1}$)	% > 0.54 $\mu g.g^{-1}$	% > 1.0 $\mu g.g^{-1}$
1983	28	334	0.03	0.21	0.08	0.08	0	0
1984	29	334	0.04	0.11	0.07	0.07	0	0

Split Lake walleye.

Year	No. of fish	$\overline{FL}$ (mm)	Min. Hg ($\mu g.g^{-1}$)	Max. Hg ($\mu g.g^{-1}$)	$\overline{Hg}$ ($\mu g.g^{-1}$)	Adjusted Hg level ($\mu g.g^{-1}$)	% > 0.54 $\mu g.g^{-1}$	% > 1.0 $\mu g.g^{-1}$
1972	30	414	0.22	1.07	0.47	0.45	20	3
1973	7	429	0.35	1.01	0.75		86	29
1974	19	448	0.17	0.97	0.54	0.38	53	0
1976	47	472	0.28	1.28	0.71	0.60	77	9
1978	66	446	0.24	2.67	0.63	0.56	56	11
1979	6	428	0.17	0.43	0.32	0.29	0	0
1980	15	493	0.27	0.82	0.52	0.30	40	0
1981	11	517	0.44	1.67	0.93	0.48	73	45
1982	13	392	0.47	1.07	0.66	0.67	62	8
1983	59	380	0.24	1.02	0.56	0.55	53	3
1984	68	355	0.08	0.89	0.38	0.43	16	0

Split Lake northern pike.

Year	No. of fish	$\overline{FL}$ (mm)	Min. Hg ($\mu g.g^{-1}$)	Max. Hg ($\mu g.g^{-1}$)	$\overline{Hg}$ ($\mu g.g^{-1}$)	Adjusted Hg level ($\mu g.g^{-1}$)	% > 0.54 $\mu g.g^{-1}$	% > 1.0 $\mu g.g^{-1}$
1972	64	581	0.12	0.88	0.41	0.37	17	0
1973	33	767	0.19	1.32	0.58	0.43	55	3
1976	101	722	0.16	1.14	0.59	0.38	57	3
1978	5	749	0.20	0.94	0.49	0.27	20	0
1979	5	827	0.63	1.00	0.79	0.32	100	0
1981	25	703	0.26	1.54	0.68	0.38	56	12
1982	19	618	0.18	1.21	0.62	0.50	53	16
1983	53	550	0.10	1.19	0.45	0.40	32	4
1984	56	520	0.07	0.86	0.31	0.31	13	0

Stephens Lake.

Year	No. of fish	$\overline{\mathrm{FL}}$ (mm)	Min. Hg ($\mu g.g^{-1}$)	Max. Hg ($\mu g.g^{-1}$)	$\overline{\mathrm{Hg}}$ ($\mu g.g^{-1}$)	Adjusted Hg level ($\mu g.g^{-1}$)	% > 0.54 $\mu g.g^{-1}$	% > 1.0 $\mu g.g^{-1}$
					Whitefish			
1983	20	398	0.07	0.45	0.24	0.18	0	0
1984	40	352	0.07	0.54	0.21	0.19	0	0
					Walleye			
1981	5	558	1.39	2.24	1.89		100	100
1983	7	465	0.97	2.60	1.78	1.36	100	89
1984	68	344	0.12	2.22	0.70	0.81	49	22
					Northern Pike			
1983	45	610	0.37	3.66	1.41	1.05	84	76
1984	44	675	0.30	3.30	1.47	0.87	88	74

APPENDIX IV: A listing of authors of previous studies on mercury in fish and the aquatic environment related to several subject headings.

Behaviour of mercury in water and sediments.

Abelson (1970)
Akagi *et al*. (1979)
Bisogni and Lawrence (1975)
Colwell and Nelson (1974)
Cranston and Buckley (1972)
Fagerstrom and Jernelöv (1971)
Furutani and Rudd (1980)
Hamby and Post (1985)
Jernelöv (1972a)
Jonasson (1970)
Jones *et al*. (1986)
Krenkel (1973, 1974)
Kudo *et al*. (1975)
Kudo *et al*. (1977)
Langley (1973)
Marmorek *et al*. (1984)
Miller and Akagi (1979)
Moore and Ramamoorthy (1984)
Mortimer and Kudo (1975)
Ramamoorthy and Kushner (1975)
Ramamoorthy and Rust (1976)
Ramamoorthy *et al*. (1977)
Ramamoorthy *et al*. (1983)
Reimers and Krenkel (1974)
Reimers *et al*. (1974)
Rust and Waslenchuk (1974)
Spangler *et al*. (1973)
Thomas (1972, 1974)
Townsend *et al*. (1974)
Turner and Lindberg (1978)
Wright and Hamilton (1982)

Mercury in fish and mercury uptake by fish.

Armstrong and Scott (1979)
Bache et al. (1971)
Benson et al. (1976)
Bishop and Neary (1974)
Bligh (1970, 1972)
Boudou and Ribeyre (1985)
de Frietas et al. (1974)
Fagerstrom et al. (1974)
Gillispie (1972)
Gillespie and Scott (1971)
Hasselrot (1968)
Håkanson (1980)
Hildebrand et al. (1980)
Huckabee et al. (1978)
Jensen and Jernelöv (1969)
Jernelöv (1972b)
Jernelöv and Lann (1971)
Johnels et al. (1967)
Karbe et al. (1975)
Kleinart (1972)
Kramer and Neidhart (1975)
Kudo (1976)
Lockhart et al. (1972, 1973)
MacCrimmon et al. (1983)
Mathers and Johansen (1985)
McFarlane and Franzin (1980)
Moore and Sutherland (1980)
Norstrom et al. (1976)
Olsson (1976)
Olson et al. (1975)
Olson and Fromm (1973)
Phillips (1977)
Phillips and Buhler (1978)
Phillips and Gregory (1979)
Pradhu and Hamdy (1977)
Reinert et al. (1974)
Rodgers and Beamish (1981, 1982, 1983)
Rodgers and Qadri (1982)
Rucker and Amend (1969)
Rudd et al. (1980)
Scott (1974)
Scott and Armstrong (1972)
Sharpe et al. (1977)
Skurdal et al. (1985)
Suns et al. (1985)
Tsai et al. (1975)
Uthe (1972)
Uthe et al. (1973)
Uthe and Bligh (1971)
Westöö (1973)
Williams and Weiss (1973)
Wobeser et al. (1970)
Wren and MacCrimmon (1973)
Zitko et al. (1970)

Mercury contamination of fish in reservoirs.

Abernathy and Cumbie (1977)
Abernathy *et al*. 1985.
Bodaly and Hecky (1979)
Bodaly *et al*. (1984a)
Bodaly *et al*. (1984b)
Boucher and Schetagne (1983)
Cox *et al*. (1979)
Garcia and Kidd (1979)
Kent and Johnson (1979)
Knight and Herring (1972)
Lodenius *et al*. (1983)
Marmorek *et al*. (1984)
McGregor, G. W. G. (1980)
Meister *et al*. (1979)
Phillips *et al*. (1980)
Potter *et al*. (1975)
Smith *et al*. (1975)

Acute and chronic effects of mercury

Anderson (1948)
Armstrong (1979)
Hara *et al*. (1976)
Harriss *et al*. (1970)
Jackim *et al*. (1970)
MacLeod and Pessah (1973)
Marshall *et al*. (1981)
McKim *et al*. (1976)
Murty (1986)
Wobeser (1975a,b)

Effects of selenium on mercury

Jones *et al*. (1986)
Marmorek *et al*. (1984)
Mikac *et al*. (1985)
Rudd and Turner (1983a,b)
Rudd *et al*. (1980, 1983)
Sandholm *et al*. (1973)
Speyer (1980)
Turner and Rudd (1983)
Turner and Swick (1983)

PROJECT 3.4

MERCURY IN HAIR OF NATIVE PEOPLE

Compiled by Manitoba Region
Medical Services Branch
Health and Welfare Canada

SPECIAL NOTE

This report was prepared for the Canada-Manitoba Agreement on the Study and Monitoring of Mercury in the Churchill River Diversion. Any conclusions, recommendations, or opinions expressed herein are those of the authors and are not necessarily the same as those of the Steering Committee for this Agreement.

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Prepared by: Prof. T.W. Clarkson
University of Rochester

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EXECUTIVE SUMMARY

Medical Services Branch of Health and Welfare Canada initiated a program in 1976 to monitor mercury levels of native residents of Manitoba. The Northern Flood Agreement area has been offered community sampling at least five times in the last 10 years. Presently, following claim #12 Arbitration and the poor turnout for sampling, annual sampling is offered to the Northern Flood Communities including South Indian Lake, Nelson House, Split Lake, York Landing, Cross Lake and Norway House. This work has been incorporated into the Canada/Manitoba Mercury Agreement as Project 3.4 - Mercury in Hair of Native People. As part of the Agreement, reports of the data, progress and results will be made to the Steering Committee. The following report summarizes the monitoring to date as well as explaining the health effects and the sampling program.

The mercury monitoring program is not a research project, but rather is set up to identify "at risk" individuals. The preventative approach is taken as there is still no specific medical treatment for the effects of methylmercury poisoning. Health education in the form of appropriate dietary advice depending on mercury levels is given with the results of the mercury testing. Information is available from the Community Health Representatives (CHR), Nursing staff, and the Zone Nutritionist.

The results of 4,668 tests, show 3,827 (82.%) in the "normal" range, 833 (17.8%) in the "increasing risk" range and 8 (0.2%) in the "at risk" range. Subsequently, re-tests have shown that those in the "at risk" range have declined and as of 1985 there were no results in the "at risk" category.

Considering mercury in all the samples, it appears that differences exist between communities. Mercury levels were highest in South Indian Lake, followed by Nelson House. York Landing and Split Lake had intermediate values. Cross Lake and Norway House had the lowest. Downward trends over the 1976 to 1986 period were seen in two communities. The six communities had generally lower mercury levels as compared to communities in Ontario and Quebec that were known to have some of the highest mercury levels in those provinces.

This report gives special consideration to females, particularly those of child-bearing age, as the pre-natal period is the most sensitive stage of the life cycle to methylmercury. Considering samples from all females, there appears to be a seasonal trend in blood mercury concentrations. The highest values are seen in the summer months and the lowest in mid-winter. The hair mercury concentrations were highly correlated with the corresponding blood levels when adjusted for the position along the hair length. The hair to blood concentration ratio is consistent with the conversion factor used by Medical Services Branch. A number of females of child-bearing age had blood mercury concentrations of 20 ppb - a value generally regarded as the upper limit of safety. The fraction above this limit differed between the six communities. Of the 73 cord blood samples tested, only 2 were in excess of 20 ppb. The actual degree of risk of pre-natal poisoning cannot be determined from the figures but the low cord blood values and the fact the excursions above the 20 ppb limit were not great, would suggest that the risk is low.

1.0 INTRODUCTION

Project 3.4, Mercury in the Hair of Native People, is not funded under the Canada/Manitoba Mercury Agreement. Rather, it is an ongoing initiative under the Environmental Contaminants Program of Medical Services Branch of the Department of National Health and Welfare. An interim order of Claim Number 12 of the Northern Flood Agreement called for extensive sampling for people of the Northern Flood Communities. This sampling has been incorporated into the mercury monitoring program of the Manitoba Region. Northern Flood Communities are now offered annual sampling while the rest of the communities are sampled on a three year rotating basis.

The work undertaken by this project was done for the express purpose of advising the residents on mercury levels and how to reduce them and not for scientific reasons. The data is not conducive to vigorous analysis as two main groups were addressed, those with possible high levels (heavy fish eaters) and those of most concern to health (women of child bearing age). Any conclusions presented must therefore be regarded as tentative as many factors could affect the data analysis ... non-random sampling, numbers of repeat samples, seasonal changes... etc... However, this report attempts to summarize the monitoring to date as well as providing some information on the health effects and the sampling program.

There is still no specific medical treatment for the effects of methylmercury, and health programs must, therefore, continue to concentrate on identifying those who may be "at risk" and on a preventative approach in the form of health education to help individuals avoid taking in a significant amount of mercury. In most cases, mercury is taken in the form of food, particularly fish and sea mammals and, therefore, advice must concentrate on dietary modification. This must be carefully done with full involvement of the native people concerned since the food sources concerned are often a staple part of the diet and intimately linked with the lifestyles of the people.

Areas and communities are being identified "at risk" where:

(a) individuals have been found to have mercury levels in excess of 100 ppb blood (or 30 ppm in hair) or,

(b) fish, in areas fished by the community have had levels in excess of 0.2 ppm. The level of 0.2 ppm is used rather than the 0.5 ppm Canadian limit because native people on reserve are liable to consume large quantities of fish. (Wheatley B, 1979).

Surveillance Programs are continuing in the "at risk" areas to identify those individuals who may be "at risk" and to follow those already identified, so as to assist both groups to modify their diets and keep their mercury levels within acceptable limits. The surveillance program will include annual hair sampling of all individuals previously noted to be in the "at risk" group.

Health education programs will be continued in the "at risk" areas and intensified where it is evident that "at risk" levels are still being reached. This includes explaining the significance of the mercury levels found in the hair, and providing advice on eating habits. A knowledge of mercury levels in local fish food sources is essential for the successful production of local solutions. Medical Services Branch is, therefore, continuing to distribute this information obtained from federal and provincial Department of Fisheries. Extensive sampling in the Northern Flood Agreement area have been conducted as part of the Canada/Manitoba Mercury Agreement.

Concern for the health effects of environmental mercury began when problems of mercury pollution were identified in northwest Ontario and northwest Quebec in the early 1970's. Subsequent recognition that environmental mercury may also pose a significant health problem in areas not directly subject to industrial pollution led to the development of an extensive program across Canada.

Medical Services Branch was involved in the initial investigations of blood mercury levels in Indian residents of northwest Ontario and northwest Quebec in the early 1970's. In 1974, it was recognized that a more intensive program was needed on Grassy Narrows and Whitedog Reserves and in northwest Quebec. Subsequently, in 1975, it was decided, in view of the multiple potential sources of environmental mercury across Canada, that the mercury program should be expanded to cover all communities for which Medical Services Branch has health responsibilities. The program was developed within the Manitoba Region in 1976.

The review and findings of the Medical Services Branch mercury program across Canada, as described in "Methylmercury in Canada - Exposure of Indian and Inuit Residents to Methylmercury in the Canadian Environment" which was released in March 1980, was supported by the findings of the McGill University Epidemiology study in Northern Quebec, the findings of which were released in April 1980. Both documents report the conclusion that severe methylmercury poisioning has not been found in Canada. However, milder forms of mercury poisoning, although difficult to prove conclusively, quite possibly are occuring in certain areas. There is particiular concern for prenatal exposure to methylmercury, since recent research has indicated that the unborn child is more sensitive to methylmercury and may be affected by maternal mercury levels in the range found in some Indian and Inuit women of child bearing age in Canada. The McGill studies suggest that pre-natal effects are occuring below 30 ppb in blood. A preliminary report from New Zealand indicates the same (Kjeldstrom et al, 1986).

2.0 HEALTH EFFECTS

This section reviews some of the major methylmercury poisonings which have occurred and then proceeds through the Canadian experience. The effects on humans, specifically children is mentioned as well as the signs and symptoms of methylmercury poisoning. The results of clinical examinations are reviewed, as well as the Canadian exposure pattern and the resulting Medical Services Branch reporting standards.

The health hazards of inorganic mercury compounds have been known for thousands of years; however, the effects of organic mercury compounds, in particular methylmercury, the primary cause of present concern, have been defined only in recent years.

Although originally described in 1940 as an industrial occupational hazard (Hunter et al, 1940; Hunter and Russell, 1954), methylmercury as a significant environmental contaminant only came to world attention in the 1960's following major epidemics of poisoning in Japan (Tsubaki et al, 1967; Katsuna, 1986). The first of these disasters followed consumption by local residents of fish contaminated by the industrial release of methylmercury into Minamata Bay. The result was that by November 1975, 143 residents had died and 899 had been diagnosed as having "Minamata Disease", a new term coined for the severe effects of methylmercury poisoning which previously had been referred to as the Hunter-Russell Syndrome (Irukayama, 1976). A second similar epidemic occurred at Niiagta in 1965, and resulted in 520 diagnosed cases and 44 deaths by the end of 1974 (Tsubaki et al, 1986). Both these epidemics were due to sustained intake of fish containing high levels of methylmercury, the result of direct industrial mercury pollution of waterways.

Other outbreaks of methylmercury poisoning have been reported from many parts of the world following the accidental consumption of seed grain treated with fungicides containing methylmercury or ethylmercury (World Health Organization, 1974). The largest and best documented outbreak took place in 1971-72 in Iraq and involved more than 6000 hospital admissions and 450 hospital deaths (Bakir et al, 1973). Unlike the Japanese experience, the Iraqi poisoning was an acute episode of short duration, with a wide range of daily intakes, following the consumption of bread made from the treated seed grain.

In Canada, a somewhat different situation prevails. Considerable quantities of inorganic mercury exist in the environment, the result of natural background contamination associated with rock and mineral deposits, industrial pollution due to old mine workings, particularly tailings from gold mines, or more recent industrial pollution. Of particular note are the considerable quantities of inorganic mercury which, between 1961 and 1970, were discharged into certain river systems by chlor-alkali plants associated with the pulp and paper industry (Sherbin, 1979).

It had been presumed for many years that, in contrast to methylmercury, inorganic mercury was relatively inert and harmless in the environment. However, in 1966 Westoo demonstrated that mercury in Swedish fish was almost entirely in the form of methylmercury, which was always the predominant compound of mercury in fish, regardless of the form of mercury contaminating the water system. Jensen and Jernelov (1969) then demonstrated that inorganic mercury is methylated by micro organisms in river and lake sediments. The methylmercury produced in this manner then undergoes bioaccumulation up the food chain and eventually reaches significant levels, especially in predatory fish and sea mammals which are commonly a food source for Native people in Canada.

This danger was recognized in 1970 and prompted the initiation of investigation into mercury levels in fish and Native people living downstream from chlor-alkali plants in northwestern Ontario and northwestern Quebec. These areas are remote and inhabited mainly by Indian groups who depend on fish as a major source of food and for whom a major component of their culture revolves around the river systems. Fish and fish products are the dominant source of mercury in food mainly in the form of methylmercury compounds.

Intake of methylmercury is reflected in elevated levels in whole blood and in erythrocytes (approximately 90% of blood methylmercury is in the red cells). Long-term fish consumption virtually determines the concentrations of methylmercury and usually total mercury in blood. Thus reference values must take into account fish consumption. In communities with high consumption rates, individuals with long-term intakes of 200 ug Hg/day will have blood levels in the range of 200 ppb (Clarkson, 1985).

Hair concentrations of methylmercury are proportional to the blood concentrations at the time of formation of the hair strand. Generally speaking, the concentration in hair is 300 times the simultaneous concentration in blood. Once incorporated into the hair strand, the concentration of mercury in hair remains unchanged. Thus longitudinal measurement of Hg in hair provides a recapitulation of methylmercury levels in blood. Airey (1983) has presented a comprehensive evaluation of mercury levels in hair. He found that mean hair Hg concentrations for a group that consumed fish once or less a month was 1.4 ug Hg/g, for once every two weeks was 1.9 ug Hg/g, once a week was 2.5 ug Hg/g and once or more a day was 11.6 ug Hg/g. These data support the conclusion that long-term fish intake determines levels of methylmercury in hair and also in blood (Clarkson, 1985).

The methylmercury in the diet is almost completely absorbed into the bloodstream and then distributed to all organs and tissues. The nervous system is the principal if not the only target tissue for effects of methylmercury in adult humans. The sensory, visual, auditory and coordinative functions, especially cerebellar, are most commonly affected (Clarkson, 1985).

Observations both on human subjects and on animals indicate that the developing central nervous system is more sensitive to damage from methylmercury than the adult nervous system. The first indications come from the outbreak of methylmercury poisoning in Minamata, Japan in the 1950's, when it was found that mothers who were slightly poisoned gave birth to infants with severe cerebral palsy. Subsequent studies on animals confirm the increased sensitivity of the fetus. The Iraq outbreak in 1971-72 also reported cases of severe damage to the central nervous system in infants prenatally exposed. More recent follow-up studies in Iraq have indicated a milder syndrome at lower exposure levels. In fact, it has been possible to demonstrate a relationship between the maximum hair level in the mothers during pregnancy and the frequency of abnormalities in their infants (Clarkson, 1985).

In summary, the clinical and epidemiological evidence indicate that the fetus is more sensitive to the toxic effect of methylmercury than are adults.

Methylmercury poisoning in adults has several important features: (1) a long latent period usually lasting several months; (2) damage almost exclusively limited to the nervous system, especially the central nervous system; (3) areas of damage to the brain being highly localized (focal), e.g. in the visual cortex, and the granular layer of the cerebellum especially in the infolded regions (sulci); (4) effects in the severe cases being irreversible due to the destruction of neuronal cells; (5) the earliest effects being non-specific subjective complaints such as paresthesia, blurred vision and malaise (Clarkson, 1985).

It is very difficult to distinguish the signs and symptoms of methylmercury poisoning, except at very high levels of exposure, because they are easily attributable to a number of other causes. These signs and symptons include: constriction of visual field, disturbance of superficial and deep sensation, lack of coordination, impairment of speech, hearing and gait (Tsubaki et al, 1977). According to the World Health Organization (1976), these signs and symptoms could appear in a mild state in the most sensitive individuals at blood methylmercury levels of 200-500 parts per billion.

As has been mentioned, all neurological signs and symptoms which are produced by methylmercury may be due to other causes. The differential diagnosis of methylmercury poisoning, particularly at the lower levels of exposure, includes polyneuritis of any origin (including vitamin deficiency, diabetes, alchoholism, pernicious anaemia and primary neurological diseases), cerebo-vascular insufficiency, cervical spondylosis, retinits pigmentosa and psychogenic disorders (Swedish Expert Group, 1971).

There have been few significant findings in clinical examinations of adults with blood mercury levels above 100 ppb but below 200 ppb (or between 30 to 60 ppm in hair). On this basis, the policy of Medical Services is that individuals with levels greater than 100 ppb in blood or its hair equivalent should continue to be designated "at risk" and should continue to receive health education and advice, although clinical examination by a neurologist is no longer mandatory, if the level has never exceeded 200 ppb. The policy also provides that if the individual does not refuse to be examined, then clinical examination of individuals with levels greater than 200 ppb in blood, or its equivalent in hair, remains mandatory. Should individuals with levels over 100 ppb but under 200 ppb request clinical examinations, or should a Medical Services Branch nurse or physician suspect that there might be significant symptomatology then, of course, clinical examinations should be offered to these individuals also.

Clinical and epidemiological assessments have been made on Canadian Indian populations seasonally exposed over a long period of time to methylmercury through fish consumption. Mercury levels in blood and hair, or both, were generally lower compared with the diagnosed cases of poisoning studies in Iraq and in the Niigata epidemic in Japan. The highest level of mercury in blood recorded in Canada was 660 ppb (Wheatley, 1979).

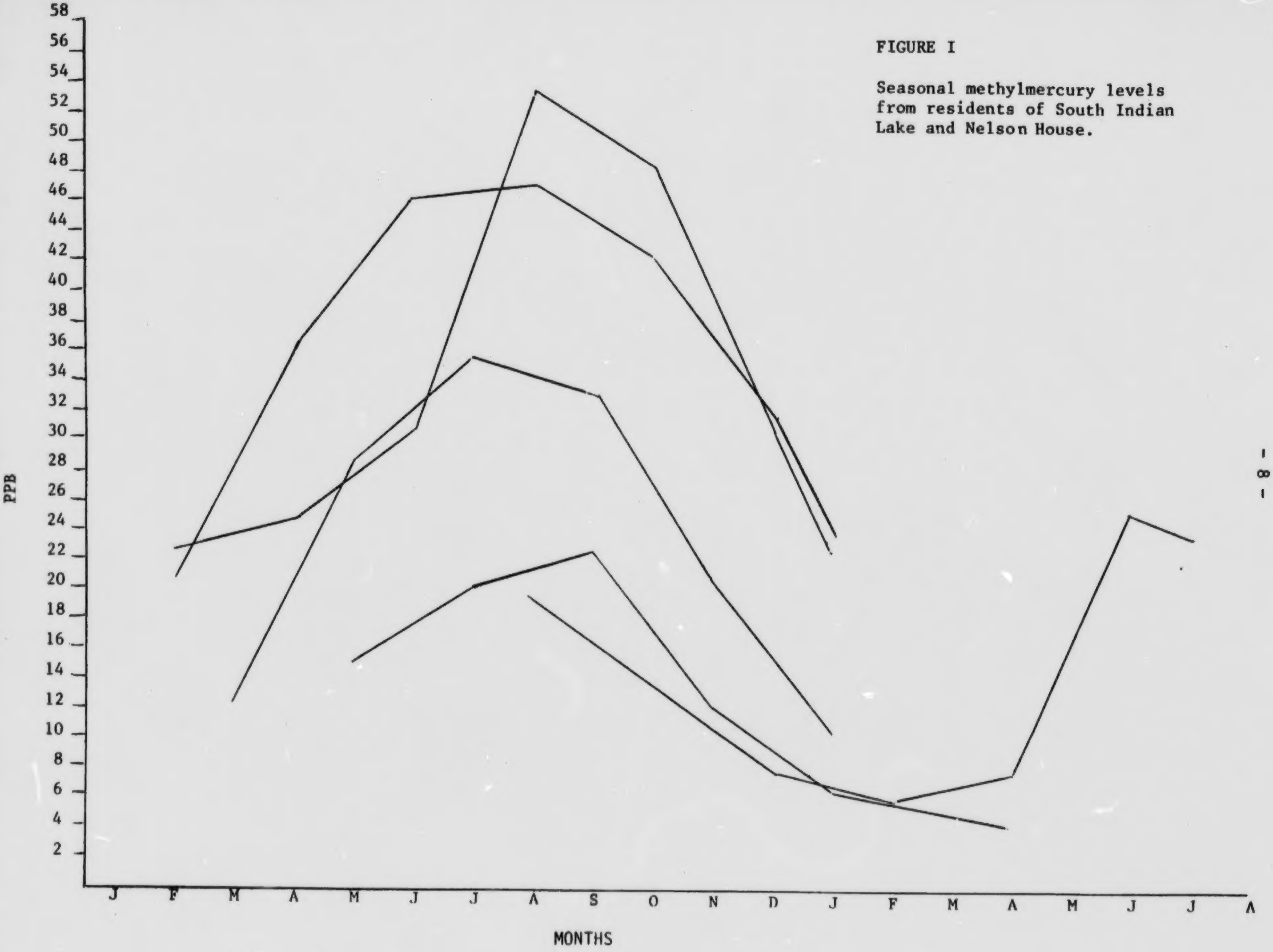
Harada et al (1976) clinically examined 89 residents of two Indian reserves (Grassy Narrows and Whitedog) who had been exposed to methylmercury by ingestion of contaminated fish. They found a number of signs and symptoms that have been associated with methylmercury intoxication. However, as the authors themselves point out, the signs and symptoms were relatively mild and many of them were thought to be due to other factors.

A report of the Medical Services Branch of the Department of National Health and Welfare, Canada (Wheatley, 1979) records the clinical examination of 84 subjects with blood mercury levels above 100 ppb. Mild symptoms and signs were found which could be attributed to possible methylmercury exposure, but the casual relationship between exposure and effects was not certain. However the 84 subjects examined, 11 cases were found where such an association could not be excluded.

As a result of analysing hair samples from northwestern Ontario and Quebec, it has been clearly demonstrated that the consumption of fish by Indian people is highest in the summer and fall; therefore, the exposure to methylmercury is on a repetitive seasonal basis, reaching a maximum in the late summer and early fall and a minimum in late winter. This exposure pattern is found in the Northern Flood Communities for people with elevated mercury levels. Figure I illustrates the seasonal exposure pattern of five residents from South Indian Lake and Nelson House.

FIGURE I

Seasonal methylmercury levels from residents of South Indian Lake and Nelson House.



This pattern of exposure to methylmercury is almost unique and is significantly different from the more sustained exposure in Japan and the acute massive exposure of many individuals in Iraq. This uniqueness has complicated assessment of the Canadian situation, but it may also have limited the severity of the health effects. For example, although the mercury levels in some species of fish in the Wabigoon-English River System in northwest Ontario have reached the same range as those seen in fish from Minamata Bay, the highest mercury significantly elevated, are an order of magnitude lower than the highest seen in Japan.

Methylmercury has an average half-life of 70 days in man. In other words, if an individual consumes fish over the summer months, and then reduces the quantity of fish eaten during the winter, the level of mercury will decrease by half approximately every 70 days until consumption increases again. It must also be noted that other variants are the methylmercury level in the food source and the amount and frequency of consumption of fish (WHO, 1976).

In 1971 a Swedish expert group reviewed the outbreaks in Japan and concluded that the lowest blood concentration associated with adverse health effects was about 200 ng Hg/ml (parts per billion). This analysis became the basis for WHO/FAO 1972 Guidelines on tolerable weekly intakes of methylmercury and for federal guidelines in Canada (Morrison, 1971). A recent review of all available data on exposed populations by a WHO expert group (1976) concluded that signs and symptoms would appear in the most sensitive individual adults at blood concentrations between 200 and 500 parts per billion (ppb), that prenatal effects may occur at lower levels, and that a safety factor of 10 should be used to define safe levels in man.

Medical Services Branch has established the following standards for reporting purposes. The hair sample mercury concentrations are converted to equivalent blood levels and reported as "parts per billion" to reduce the possibility of confusion among different units:

Below 20 ppb - normal acceptable range
20-100 ppb - increasing risk
Above 100 ppb - at risk

3.0 MERCURY SAMPLING PROGRAM

The mercury sampling program initiated in 1976 is outlined in detail in Appendix I. It consists of Community Health Nurses and Community Health Representatives (CHR) taking hair samples from community residents and forwarding the samples to the Occupational Health Unit in Ottawa. An automated mercury analysis system was set up to handle the large number of samples. The hair sample is divided into one centimetre segments up to 12 centimetres long, for monthly mercury determinations as hair grows at an average rate of one centimetre per month. In this way, a historical chart can be developed to observe a person's mercury level for up to the last 12 months. Results are returned to the Manitoba Regional Office. There the highest value is recorded and reported.

The Chief and Council receive a summary table broken down into the three ranges: 0-19, 20-99, and greater than 100 ppb. The Nurse-in-Charge receives a list of the individual results with both high and low values. The person sampled receives a letter indicating the high and low value with explanatory information. Follow-up is provided as appropriate.

The sampling program initiated in 1976 depends on a cooperative effort as all sampling is done on a volunteer basis. Initial response was not good from most Northern Flood Agreement communities including South Indian Lake and Nelson House, the two communities of most concern. In conjunction with the Canada/Manitoba Mercury Agreement and claim #12 of the Northern Flood Agreement; special public meetings were held, a video tape was produced with a Cree translation, a whitefish recipe contest and a colouring contest were held, information was supplied through the school system, all in an effort to increase knowledge of the problem and to increase the participation in the mercury sampling program. The program offering 100% sampling to the communities was met with varying degrees of success as will be shown in section 4.0.

4.0 ANALYSIS OF MERCURY RESULTS

4.1 SUMMARY

All the data were examined on mercury levels in samples of blood and hair collected from the six communities from 1976 to 1986. The samples were collected for purposes of advising the residents on mercury levels and how to reduce them and not for scientific reasons. The conclusions presented here must therefore be regarded as tentative.

Considering mercury in all the samples, it appears that differences exist between communities. Blood levels were highest in South Indian Lake, followed by Nelson House. York Landing and Split Lake had intermediate values. Cross Lake and Norway House had the lowest. Downward trends over the 1976 to 1986 period were seen in two communities. The six communities had generally lower mercury levels as compared to communities in Ontario and Quebec that were known to have some of the highest mercury levels in those provinces.

This report gives special consideration to females, particularly those of child-bearing age, as the pre-natal period is the most sensitive stage of the life cycle to methylmercury. Considering samples from all females, there appears to be a seasonal trend in blood mercury concentrations. The highest values are seen in the summer months and the lowest in mid-winter. The hair mercury concentrations were highly correlated with the corresponding blood levels when adjusted for the position along the hair length. The hair to blood concentration ratio is consistent with the conversion factor used by Medical Services Branch. A number of females of child-bearing age had blood mercury concentrations of 20ppb-a value generally regarded as the upper limit of safety. The fraction above this limit differed between the six communities. Of the 73 cord blood samples tested, only 2 were in excess of 20ppb. The actual degree of risk of pre-natal poisoning cannot be determined from the figures but the low cord blood values and the fact the excursions above the 20ppb limit were not great would suggest that the risk is low.

4.2 ALL COMMUNITIES

The main emphasis is given to the six communities in the flood agreement but a brief comparison is made with selected communities in Ontario and Quebec.

ALL COMMUNITIES

4.2.1 ALL SAMPLES

4.2.1.1 Blood mercury levels

The table overleaf summarizes the data for all blood and hair samples collected over the sampling period from 1971 to 1986. It indicates the total number of samples collected and the percentage of the population that were sampled at least once during the sampling period (Sampled, % population). The table also gives information on mercury levels. All mercury values have been converted to equivalent blood levels in ppb (parts per billion). Thus the percentage of reported samples are given exceeding specified mercury levels (> 20 ppb and > 99 ppb). A statistical estimate is also given of the minimum and maximum number of samples (columns headed "minimum %" and "maximum %") expected to lie above the generally accepted health limit of 20ppb. The statistical procedures are described by Fleiss (1981) and estimate 95% confidence limits on the percent above 20ppb. It is assumed that the proportion of samples above 20 ppb is representative of the proportion in the whole community.

Within the six communities, South Indian Lake followed by Nelson House have the highest reported mercury levels whether viewed as the percent above 20ppb or the percent above 99ppb. The statistical range for samples over 20ppb indicates that South Indian Lake is significantly higher than Nelson House. York Landing and Split Lake have intermediate levels significantly below those of Nelson House. Cross Lake and Norway House had the lowest reported values significantly below York Landing and Split Lake. Nelson House, Cross Lake and Norway House had lower percentages of their populations sampled as compared to the other three communities. It is not clear how the lower rate of sampling would affect the mercury comparisons.

The five communities in Ontario and Quebec were selected for comparison because they have some of the highest reported levels in all of Canada.These five communities differ sharply from the six Manitoba communities in the percentage of samples above 99ppb. Medical Services Branch regard 100ppb and above as "at risk" levels for adults. In terms of the percentage of samples above 20ppb, South Indian Lake and Nelson House are the only Manitoba communities that fall in the range of these five communities.

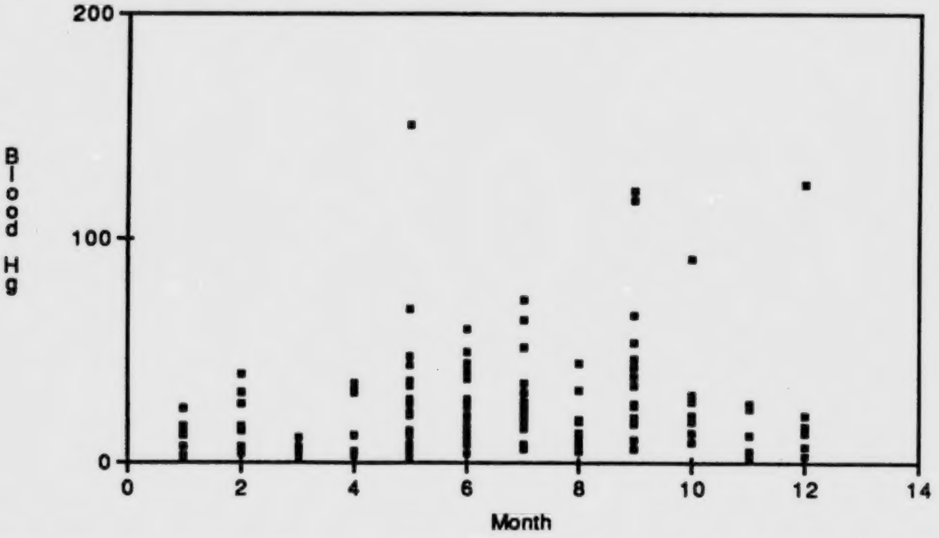
COMMUNITY COMPARISONS FOR MANITOBA, ONTARIO AND QUEBEC (1971-1986)

Community	Population	Samples number	Sampled % Population	> 20 ppb %	minimum %	maximum %	>99 ppb %
MANITOBA							
SOUTH INDIAN LAKE	655	924	89.8%	35.3%	31.6%	37.7%	0.6%
NELSON HOUSE	1331	717	33.4%	24.7%	21.3%	27.4%	0.3%
YORK LANDING	425	369	53.2%	17.6%	13.7%	21.2%	0.0%
SPLIT LAKE	1037	870	60.2%	16.7%	14.2%	19.1%	0.0%
CROSS LAKE	2079	988	39.9%	8.0%	6.3%	9.6%	0.0%
NORWAY HOUSE	2335	800	25.4%	6.0%	4.4%	7.5%	0.0%
ONTARIO AND QUEBEC							
MISTASSINI, QUE.	1733	6580	99	44.0%	42.8%	45.2%	4.0%
Waswanipi, Que.	742	2180	100	40.6%	38.5%	42.6%	5.1%
Grassy Narrows, Ont.	477	2578	100	32.6%	30.8%	34.4%	6.3%
Chisasi, Que.	1811	2040	90 - 100	29.5%	27.5%	31.4%	2.0%
Whitedog, Ont.	630	2759	85 - 100	24.0%	22.4%	25.6%	2.5%

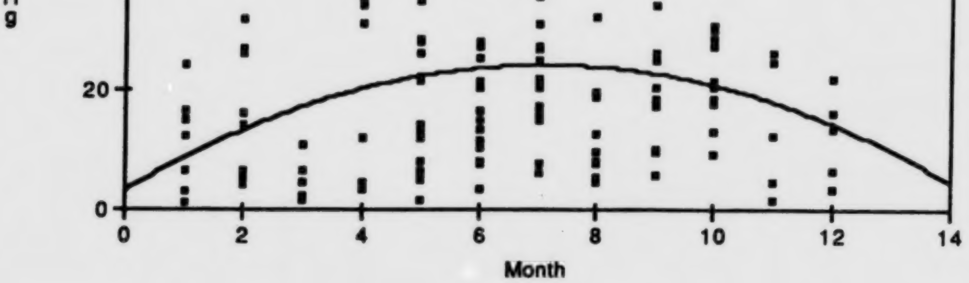
Data from tables on individual communities

4.2 ALL FEMALES

4.2.1 Seasonal changes



This figure relates mercury blood levels in repeat samples to the month when the level existed in blood. A tendency is seen for the higher levels to occur in the summer months. However five very high values, all above 90ppb, tend to obscure the seasonal effect. When these were deleted a more pronounced seasonal effect is seen as in the figure below.



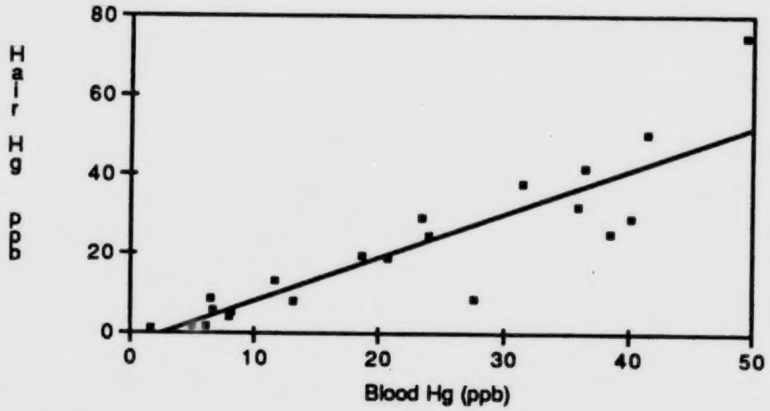
ALL COMMUNITIES

The continuous line represents a best fit to the data using a second order polynomial equation. The dependent variable, Y(blood mercury levels) is related to the independent variable, X(month) by the second order polynomial

$$Y = a_1X_1 + a_2X_2 + \text{error}$$

Details on the computation of the best fit values for the coefficients a_1 and a_2 are given by Beaton (1964)

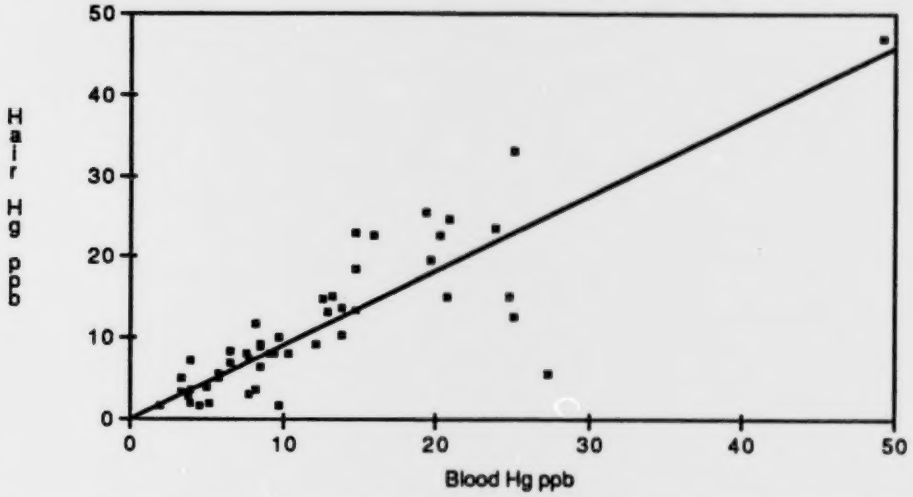
4.2.2 Hair to blood ratios



A graph of hair versus blood mercury concentrations.

The hair and blood samples were collected at the same time. The first centimeter next to the scalp was measured. The hair concentrations in parts per million (ppm) were divided by 0.3 to convert to corresponding blood concentrations in parts per billion (ppb) as described by Wheatley (1979). The line was drawn by least- squares regression analysis. The slope of the line is 1.105, insignificantly different from unity and the intercept is not different from zero.

ALL COMMUNITIES

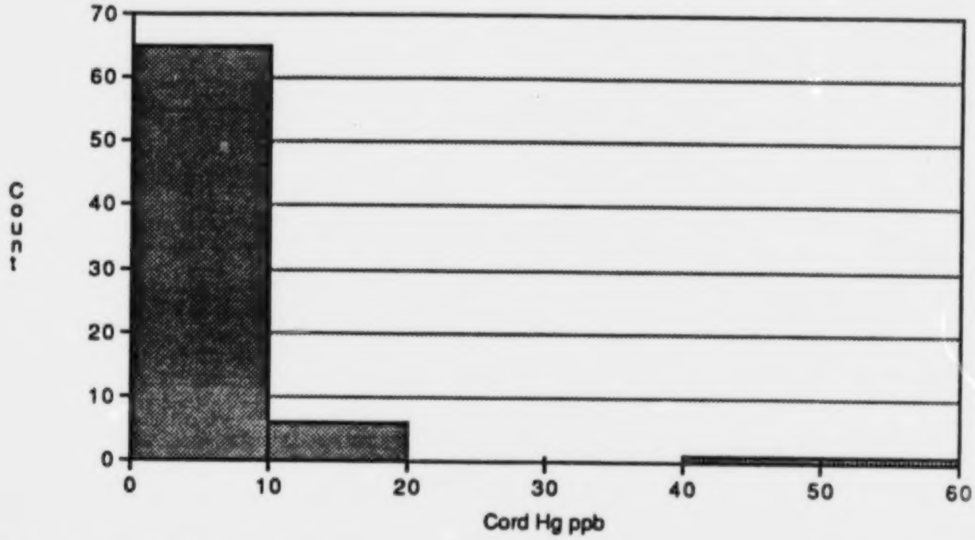


A graph of hair versus blood mercury concentrations using the same data as above but the regression line was forced through zero.

This was done on the basis that methyl mercury in the diet was the only source of mercury in hair and the intercept in the previous graph is not different from zero. The slope of the line, drawn by least-squares linear regression, is 1.01. In other words, the conversion factor for changing hair concentrations to blood concentrations using the first centimeter next to the scalp seems to work well and has been in use by Medical Services Branch since first introduced 1n the 1970's (Wheatley, 1979)

ALL COMMUNITIES

4.2.3 Cord bloods



The distribution of samples according to the concentration of mercury in cord blood in all communities.

Descriptive Statistics on cord bloods

Data File: Cords all
Variable: Cord Hg ppb   Observations: 73

Minimum: 0.700   Maximum: 51.800
Range: 51.100   Median: 4.100

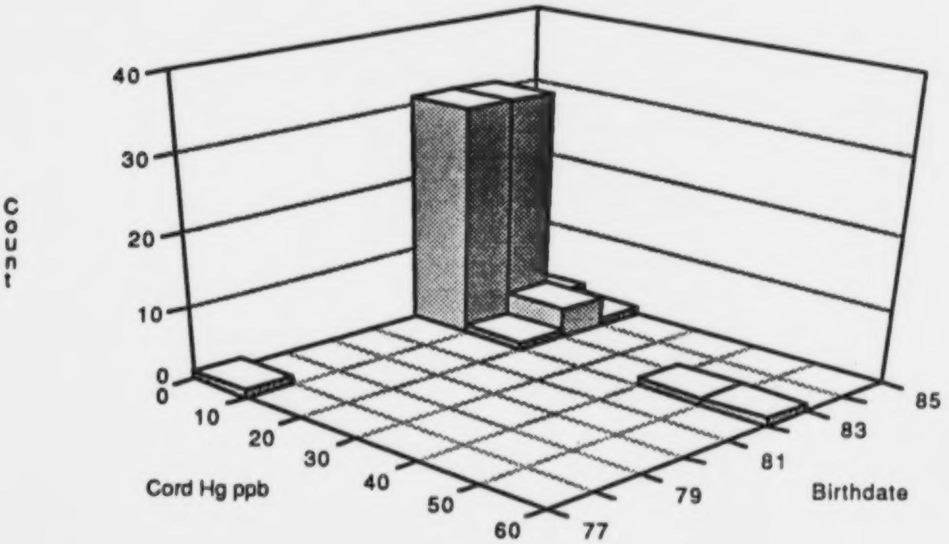
Mean: 6.225   Standard Error: 0.864

Variance: 54.443
Standard Deviation: 7.379
Coefficient of Variation: 118.537

Skewness: 4.557   Kurtosis: 23.170

The descriptive statistics confirm the visual impression from the histogram that the distribution of mercury levels is highly skewed as a result of the two high values. All other samples are below 20ppb.

ALL COMMUNITIES



This 3D-histogram indicates the two high values were in 1983 and that most of the cord bloods were collected in the period 1982 to 1985. It is therefore impossible to determine any trends with time.

4.3 INDIVIDUAL COMMUNITIES

4.3.1 South Indian Lake

SUMMARY & EVALUATION

This community has the highest mercury levels. An average of 0.6% exceeded 99 ppb and about 35% were between 20 and 99ppb. Samples were collected from females including females of child-bearing age from 1977 to 1986. The distribution of blood mercury levels was similar between females of child-bearing age as compared to all females in the community A number of samples exceeded 20ppb and the highest recorded level 125 ppb. Sixteen cord blood values were reported from an estimated total number of births of about 250. All were below 20ppb except for two samples having values between 40 to 52ppb. There was evidence for a seasonal pattern for mercury values with high values in the summer and low values in the winter. No trend in mercury levels could be detected according to the year of sampling.

The risk to adult members of the population would appear to be negligible but there is some concern that a number of females of child-bearing age exceed the generally recognized safe limit of 20ppb, that one value was 125ppb and that two cord blood values exceeded 20 ppb.

4.3.1.1 ALL SAMPLES

The table overleaf summarizes the data for all blood and hair samples collected over the sampling period from 1976 to 1986. It indicates the number of samples collected per year and the samples expressed as % of population. It also gives information on mercury levels. All mercury values have been converted to equivalent blood levels in ppb (parts per billion). Thus the number of samples are given falling in specified ranges of mercury levels. The latter are expressed as absolute numbers and as % of the samples collected. The lower part of the table gives a statistical estimate of the minimum and maximum number of samples expected to lie above the generally accepted health limit of 20ppb. The statistical procedures are described by Fleiss (1981) and estimate the 95% confidence limits for samples above 20ppb.

SOUTH INDIAN LAKE(Summary of all samples)

Date	samples number	0 -19ppb number	20 - 99ppb number	>99ppb number	Samples % population	0 - 19ppb % samples	20 - 99ppb % samples	>99ppb % samples
1977	37	29	8	0	5.6%	78.4%	21.6%	0.0%
1978	12	9	3	0	1.8%	75.0%	25.0%	0.0%
1979	100	26	68	6	15.3%	26.0%	68.0%	6.0%
1980	1	1	0	0	0.2%	100.0%	0.0%	0.0%
1981	165	102	63	0	25.2%	61.8%	38.2%	0.0%
1982	389	274	115	0	59.4%	70.4%	29.6%	0.0%
1983	59	55	4	0	9.0%	93.2%	6.8%	0.0%
1984	105	70	35	0	16.0%	66.7%	33.3%	0.0%
1985	56	31	25	0	8.5%	55.4%	44.6%	0.0%
	Total	Total	Total	Total		Overall	Overall	Overall
Total/Overall	924	597	321	6		64.6%	34.7%	0.6%

Date	Samples number	samples % population	> 20 ppb % samples	minimum % samples	maximum % samples
1977	37	5.6%	21.6%	10.4%	38.6%
1978	12	1.8%	25.0%	6.7%	57.2%
1979	100	15.3%	74.0%	64.1%	82.0%
1980	1	0.2%	0.0%		
1981	165	25.2%	38.2%	30.8%	46.1%
1982	389	59.4%	29.6%	25.2%	34.5%
1983	59	9.0%	6.8%	2.2%	17.3%
1984	105	16.0%	33.3%	24.6%	43.3%
1985	56	8.5%	44.6%	31.5%	58.4%

The population or total number of residents in this community is assumed to be 655

South Indian Lake

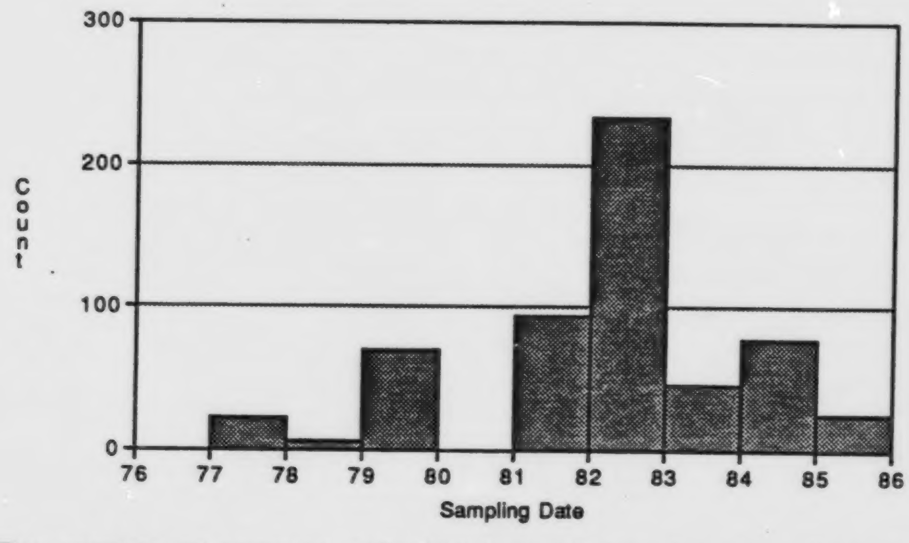
The data in the table indicates that only six out of 924 samples exceeded 99ppb. An average of about 35% of the samples were in the range 20 to 99 ppb. The 1979 levels (% above 20ppb) are significantly higher than 1976. The years after 1979 are significantly lower but no trend with time is seen after 1983.

Approximately 90% of the population were sampled at least once during the 10 year sampling period.

4.3.1.2 ALL FEMALES

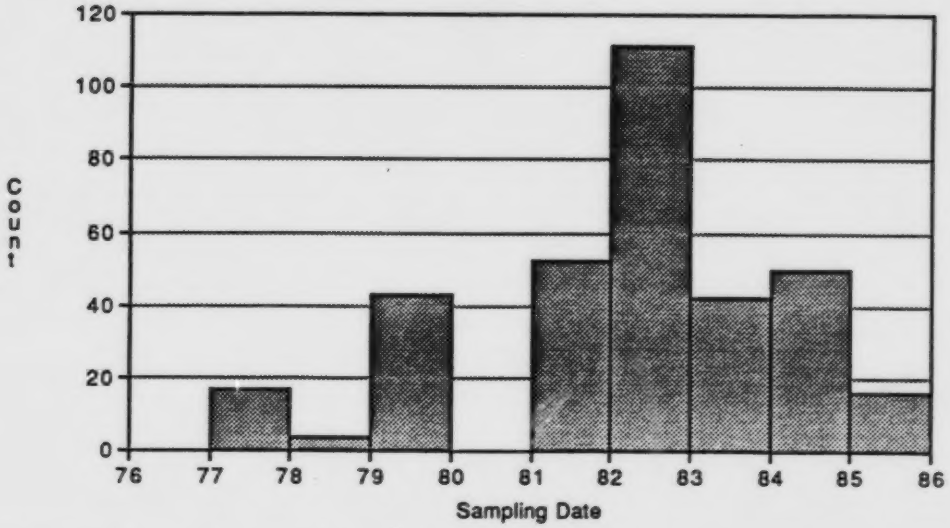
4.3.1.2.1 Sample collection

The histograms listed below summarize information on the number of samples collected in all females and also in females of child bearing age (assumed to be 15 to 45 years). Information is also summarized on the age at the time of sampling. However no correction has been made for the position of the segment on the hair sample so that both sampling date and sampling age may not correspond exactly with the actual date that the blood mercury concentration existed.

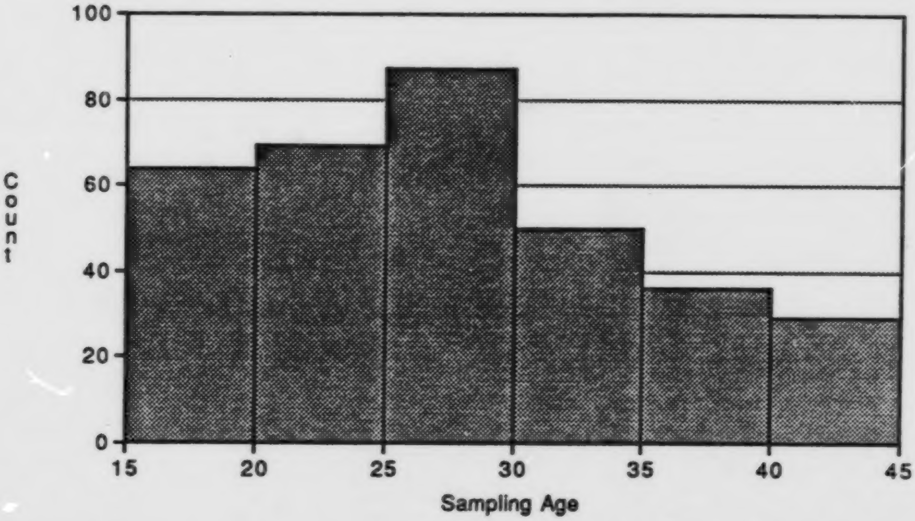


The number of samples collected from all females according to the year of sampling. A total of 579 samples were reported.

South Indian Lake

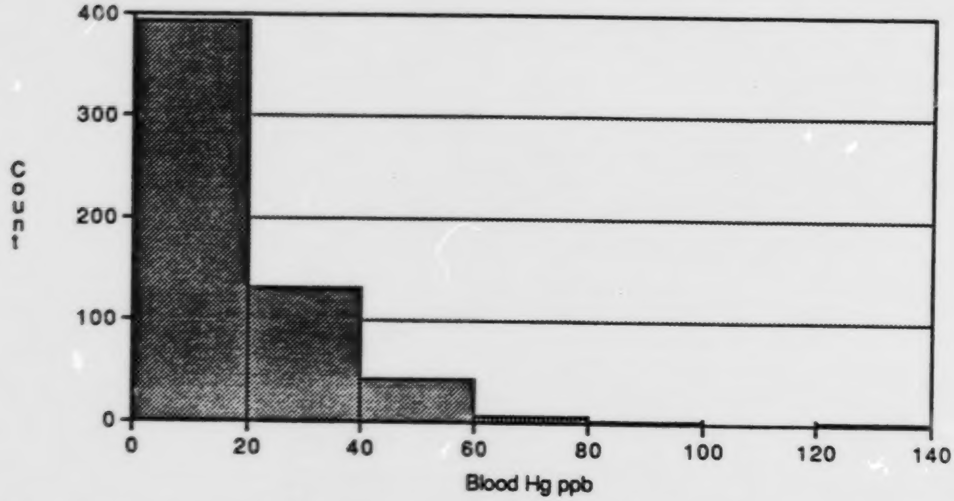


The number of samples collected (N=335) according to the date of sampling in all females of child-bearing age (15 to 45 years). The distribution of samples appears to be the same in both females of child-bearing age and all females.

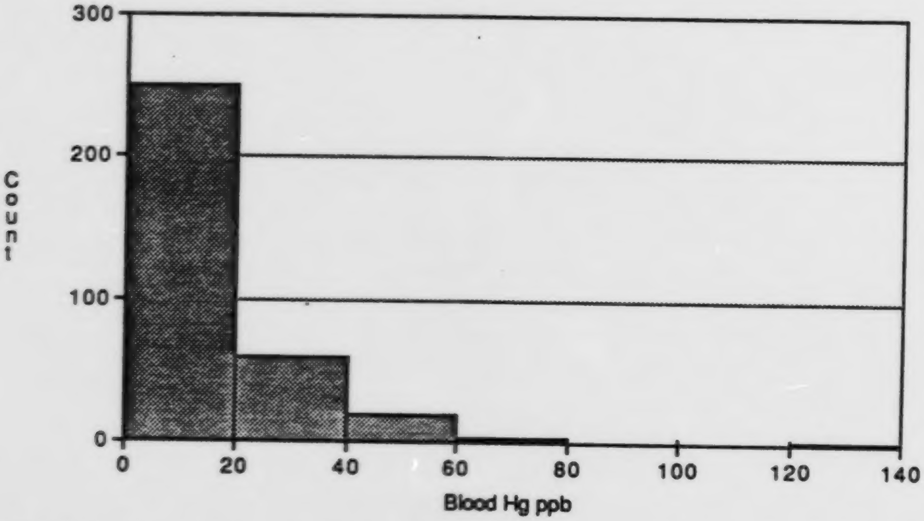


The distribution of samples according to age at the time of sampling in all females of child-bearing age (15 to 45 years). The histogram indicates that samples were collected from all age groups in the range from 15 to 45 years.

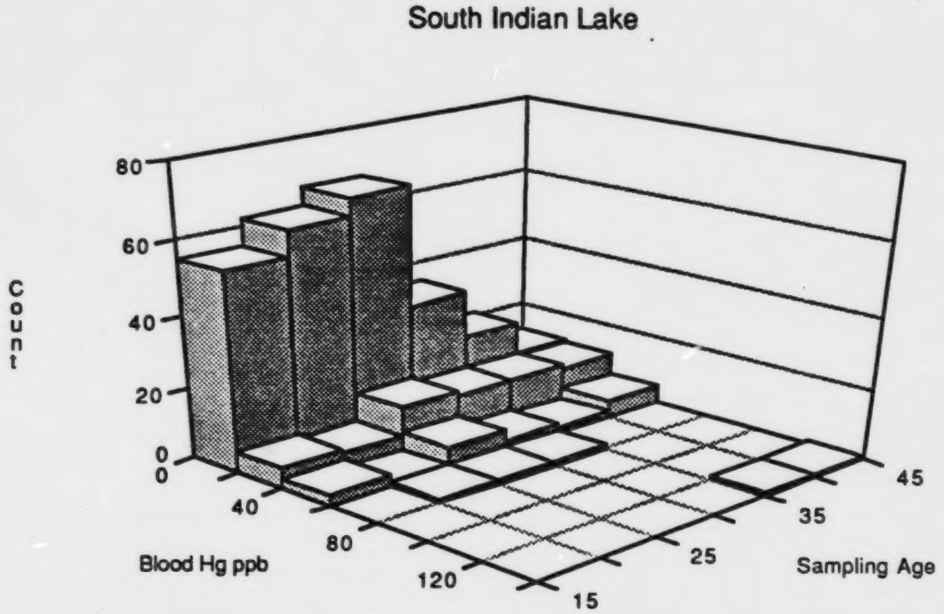
4.3.1.2.2 Blood levels



The distribution of blood mercury levels in all females sampled (N= 579). The highest reported value was 125ppb. The median blood level was 12.7 ppb.

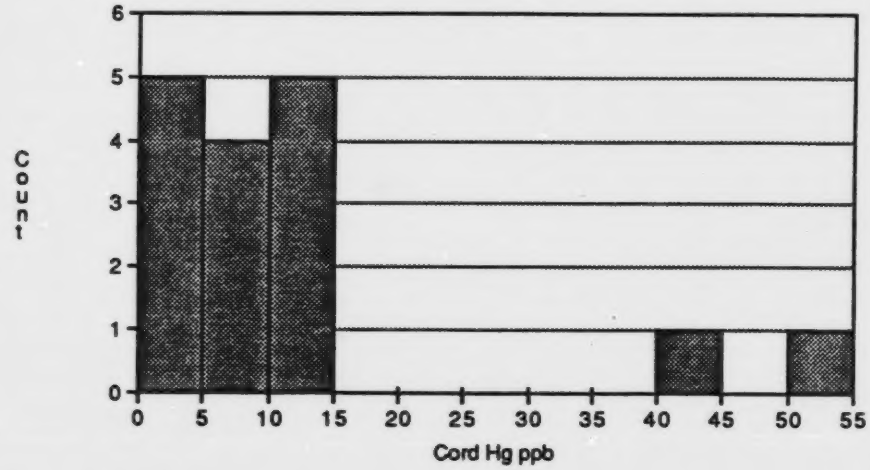


The distribution of mercury levels in females (N= 335) of child-bearing age at the time of sampling. The highest reported level was 125ppb. The median blood level was 11.3ppb. The distribution of blood levels in females of child-bearing age appears to be similar to the distribution for all females in the community.



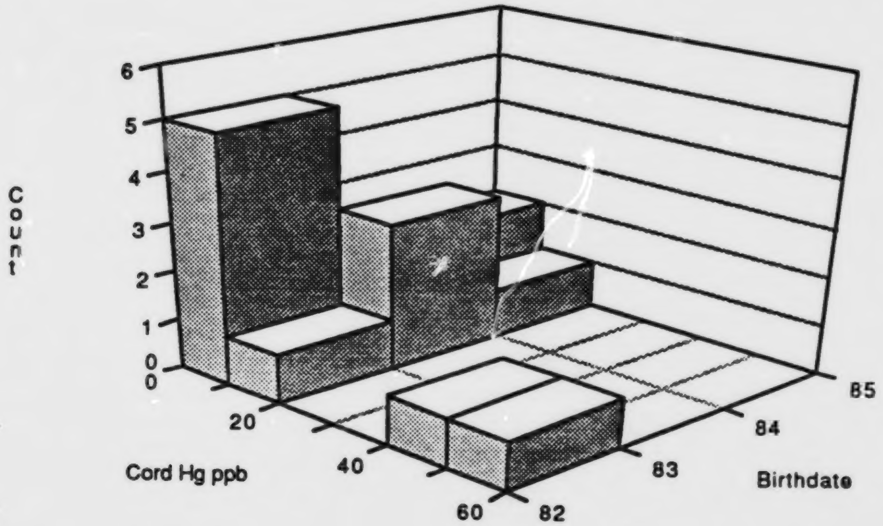
The distribution of samples according to the blood levels and the age at the time of sampling in females of child-bearing age. It may be seen that a number of samples exceed the generally accepted safe limit of 20ppb. It is of concern that a number of values are in the 40 to 80ppb range and that two values exceed 120ppb. No trend in blood levels with age is apparent except that the two highest levels were in women over 35 years of age.

4.3.1.2.3 Cord blood levels



The distribution of mercury levels in cord blood samples. Only two values were found above 20ppb. The maximum level was 52ppb and the median was 8ppb.

South Indian Lake

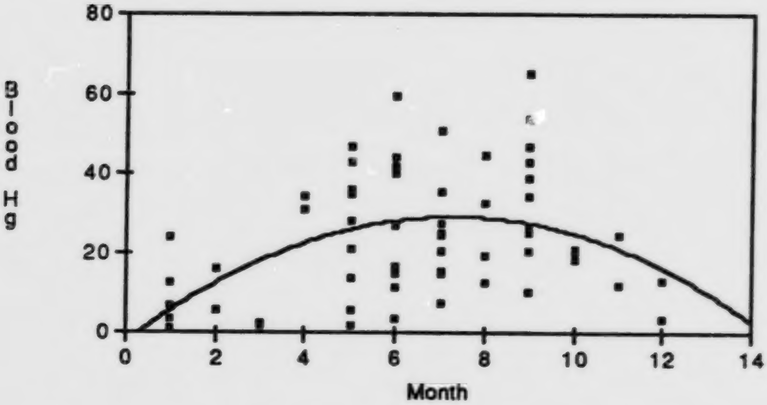


The number of samples according to birthdate and mercury levels. The two high values were in the 1982-83 period. No values were reported before 1982.

4.3.1.2.4 Repeat Samples

4.3.1.2.4.1 Seasonal Trends

The graph presented below is an analysis of repeat samples from eleven individuals to see if seasonal trends exist in mercury levels.



Blood mercury levels according to month in females having five or more repeat samples. The continuous line is a least-squares best fit to the data points using a second order polynomial equation. The dependent variable, Y(blood mercury levels) is related to the independent variable, X(months) by the second order polynomial

$$Y = a_1 X_1 + a_2 X_2 + \text{error}$$

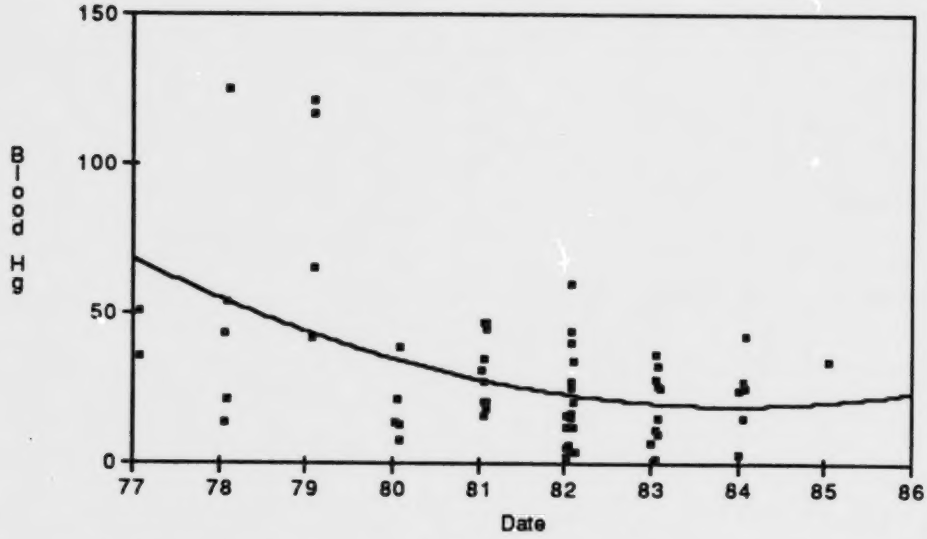
Details on the computation of the best fit values for the coefficients a_1 and a_2 are given by Beaton(1964)

The tendency to have high mercury levels in the summer months and lower values in the winter time is statistically significant. Three abnormally high points were not included in this analysis because they would have given an excessive distortion of this curve.

This seasonal variation is typical of native fish eating communities in Canada including those studied in Northwest Ontario and Quebec.

4.3.1.2.4.2. Trends by Year.

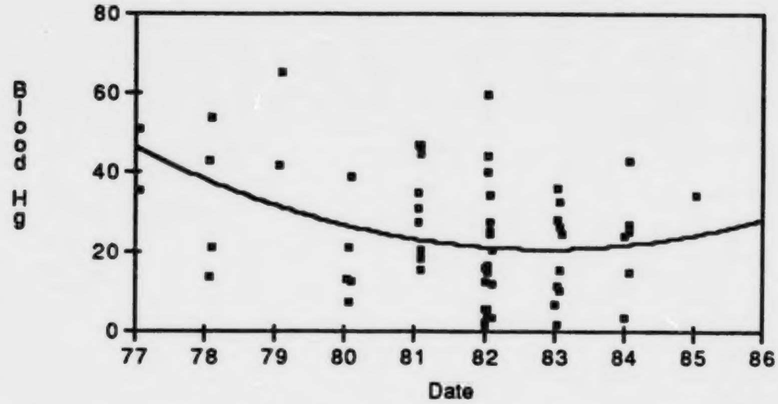
The graphs presented below are an attempt to detect yearly trends in the eleven individuals from whom five or more samples have been taken.



A graph of blood mercury levels according to the year of sampling in females having five or more repeat samples.

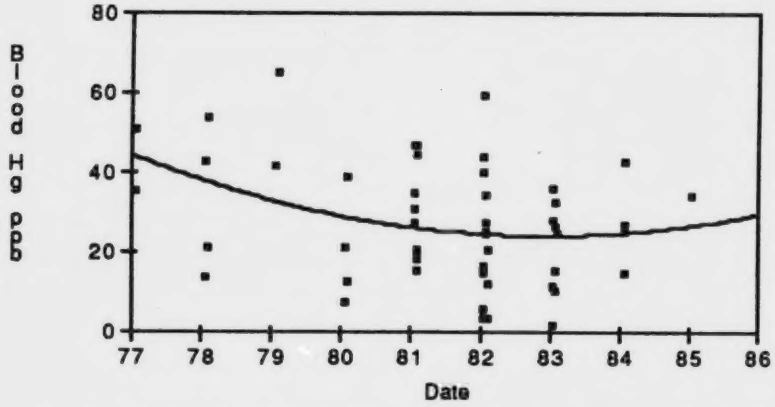
The continuous line was drawn by a least-squares fit using a second order polynomial equation as described previously in the analysis of seasonal trends. The coefficients relating blood concentrations to date of collection were not significantly different from zero (p=0.14 and 0.15) There is an apparent tendency for values to decline but three very high values may have excessively influenced the computation. These were deleted in the next graph.

South Indian Lake



A graph of blood mercury levels according to the year with the three highest values excluded. It may be seen that a slight tendency still persists for values to decline over the years. However, the coefficients relating blood concentrations to date of collection were not significantly different from zero (p=0.10 and 0.11).

This analysis may be biased by the seasonal changes. In the last figure presented below samples collected in the winter months were deleted.



Blood mercury levels according to year after data for the winter months (December to March inclusive) were deleted. Having removed seasonal effects, no tendency can be seen for mercury levels to change with the year. The coefficients relating blood concentrations to date of collection were not significantly different from zero (p=0.21 and 0.22)

4.3 INDIVIDUAL COMMUNITIES

4.3.2 NELSON HOUSE

SUMMARY and EVALUATION

This community has the second highest mercury levels. An average of 0.3% exceeded 99ppb and about 24 % were between 20 and 99ppb. Samples were collected from females including females of child-bearing age from 1977 to 1986. The distribution of blood mercury levels was similar between females of child-bearing age as compared to all females in the community A number of samples exceeded 20ppb and the highest recorded level 150ppb. In females of child-bearing age, there was a tendency for the higher blood levels to occur in the older age group. Only three results were recorded on cord bloods, all well below 20ppb Since approximately 400 births occurred over a ten year sampling period it is impossible to draw any conclusions from the small number of cord bloods. The results on females of child-bearing age, however, suggest that higher cord blood values would have been found if the sampling had been more extensive. There was a tendency for the mercury levels to fall during the sampling period.

The risk to adult members of the population would appear to be negligible but there is some concern that a number of females of child-bearing age exceed the generally recognized safe limit of 20ppb and that one value of 150ppb was found.

4.3.2.1 ALL SAMPLES

The table overleaf summarizes the data for all blood and hair samples collected over the sampling period from 1976 to 1986. It indicates the number of samples collected per year and the samples expressed as % of population. It also gives information on mercury levels. All mercury values have been converted to equivalent blood levels in ppb (parts per billion). Thus the number of samples are given falling in specified ranges of mercury levels. The latter are expressed as absolute numbers and as % of the samples collected. The lower part of the table gives a statistical estimate of the minimum and maximum number of samples expected to lie above the generally accepted health limit of 20ppb. The statistical procedures are described by Fleiss (1981) and estimate the 95% confidence limits for samples above 20ppb.

Nelson House (Summary of all samples)

DATE	Samples number	0 -19ppb number	20 - 99ppb number	>99ppb number	Samples % population	0 - 19ppb % samples	20 - 99ppb % samples	>99ppb % samples
1976	4	1	3	0	0.3%	25.0%	75.0%	0.0%
1977	59	57	2	0	4.4%	96.6%	3.4%	0.0%
1978	102	63	38	1	7.7%	61.8%	37.3%	1.0%
1979	130	71	59	0	9.8%	54.6%	45.4%	0.0%
1980	0	0	0	0	0.0%			
1981	0	0	0	0	0.0%			
1982	56	50	6	0	4.2%	89.3%	10.7%	0.0%
1983	24	24	0	0	1.8%	100.0%	0.0%	0.0%
1984	221	181	39	1	16.6%	81.9%	17.6%	0.5%
1985	121	93	28	0	9.1%	76.9%	23.1%	0.0%
	Total	Total	Total	Total	Total	Overall	Overall	Overall
Total/overall	717	540	175	2		75.3%	24.4%	0.3%

DATE	Samples number	Samples % population	> 20 ppb % samples	minimum %samples	maximum % samples
1976	4	0.3%	75.0%	21.9%	98.7%
1977	59	4.4%	3.4%	0.6%	12.8%
1978	102	7.7%	37.3%	28.1%	47.5%
1979	130	9.8%	45.4%	36.7%	54.3%
1980	0	0.0%	0.0%		
1981	0	0.0%	0.0%		
1982	56	4.2%	10.7%	4.4%	22.5%
1983	24	1.8%	0.0%		
1984	221	16.6%	18.1%	13.4%	24.0%
1985	121	9.1%	23.1%	16.1%	31.8%

The population or total number of residents is assumed to be 1331.

Nelson House

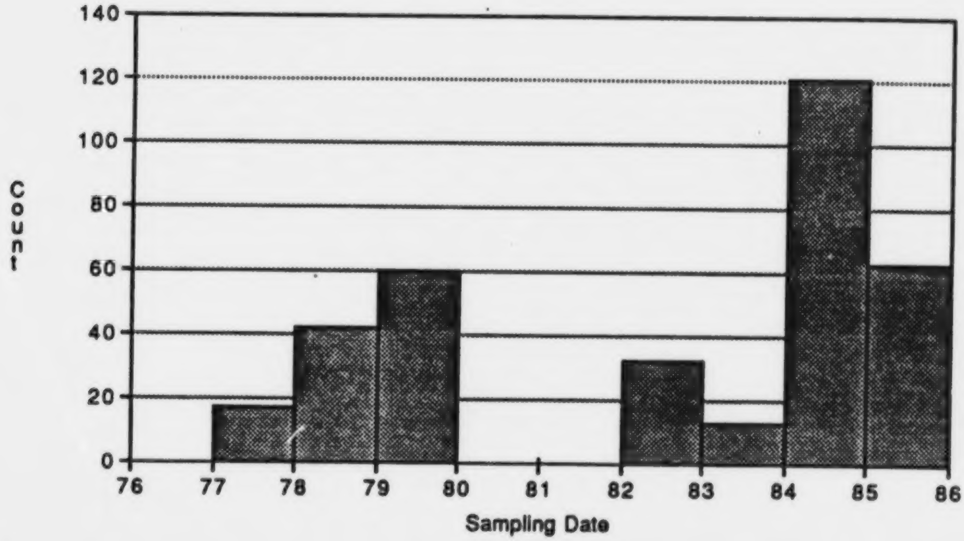
As seen in the table, only two blood levels out of 717 samples exceeded 99 ppb and an average of about 24% was in the range of 20 to 99ppb. The years 1978 and 1979 are both higher than the 1977 levels (% above 20 ppb). Levels in later years are lower. Samples collected from 1983 to 1986 are significantly lower than the 1978-79 levels.

Approximately 33% of the population were sampled at least once during the 10 year sampling period.

4.3.2.2 ALL FEMALES

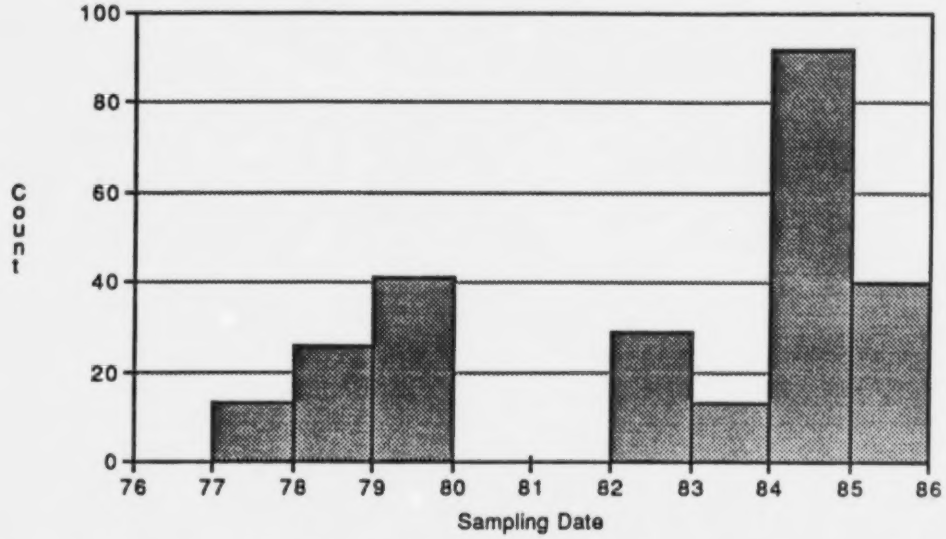
4.3.2.2.1 Sample collection

The histograms listed below summarize information on the number of samples collected in all females and also in females of child bearing age (assumed to be 15 to 45 years). Information is also summarized on the age at the time of sampling. However no correction has been made for the position of the segment on the hair sample so that both sampling date and sampling age may not correspond exactly with the actual date that the blood mercury concentration existed.

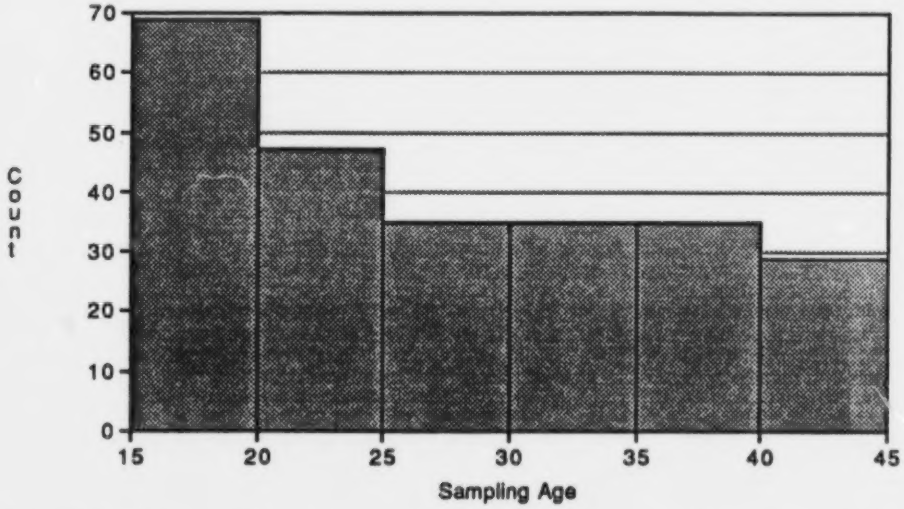


The number of samples collected from all females according to the year of sampling. The total number was 348.

Nelson House

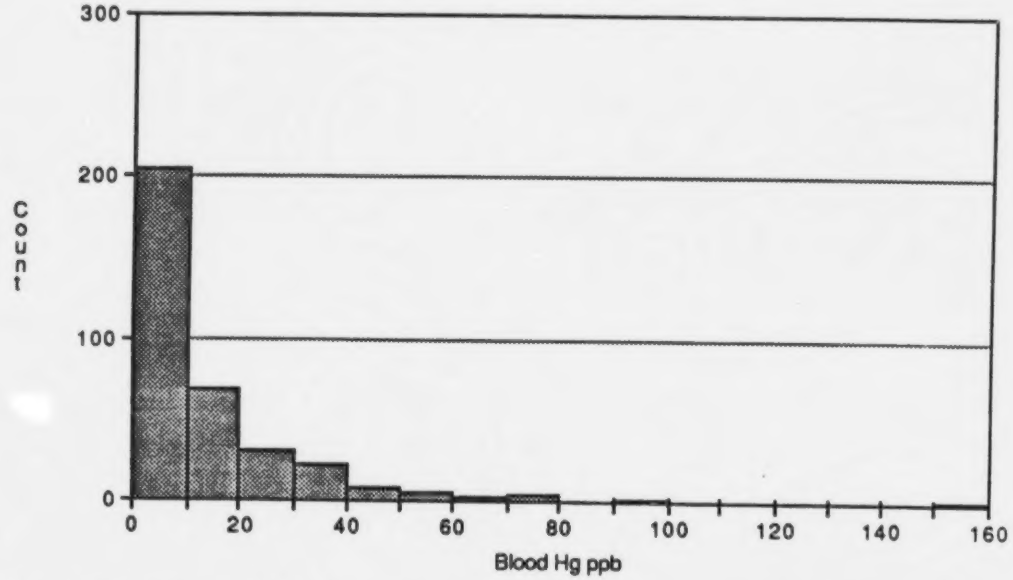


The number of samples collected according to the date of sampling in all females of child-bearing age (15 to 45 years). The distribution of samples appears to be the same in both females of child bearing age and in all females.

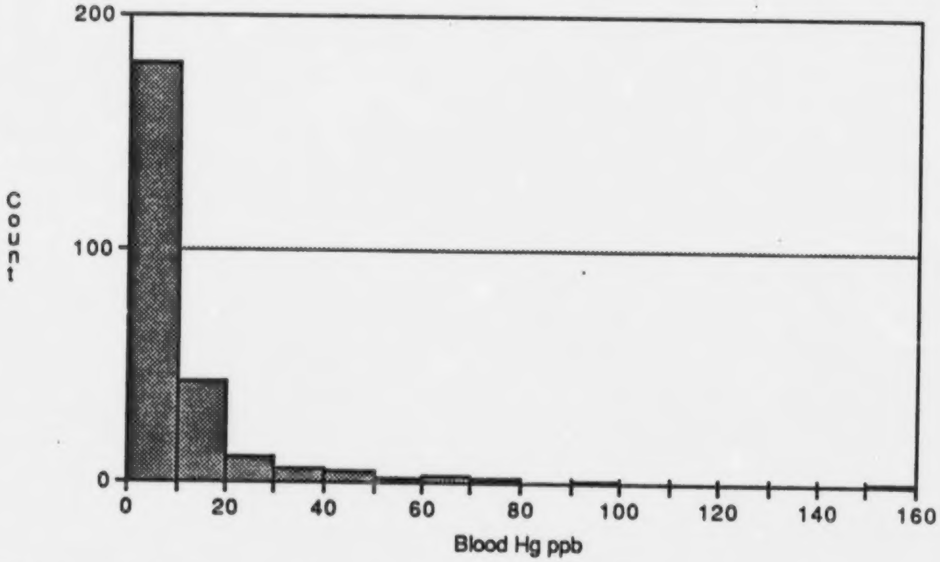


The distribution of samples according to age at the time of sampling in all females of child bearing age (15 to 45 years). The histogram indicates that samples were collected from all age groups in the child bearing range.

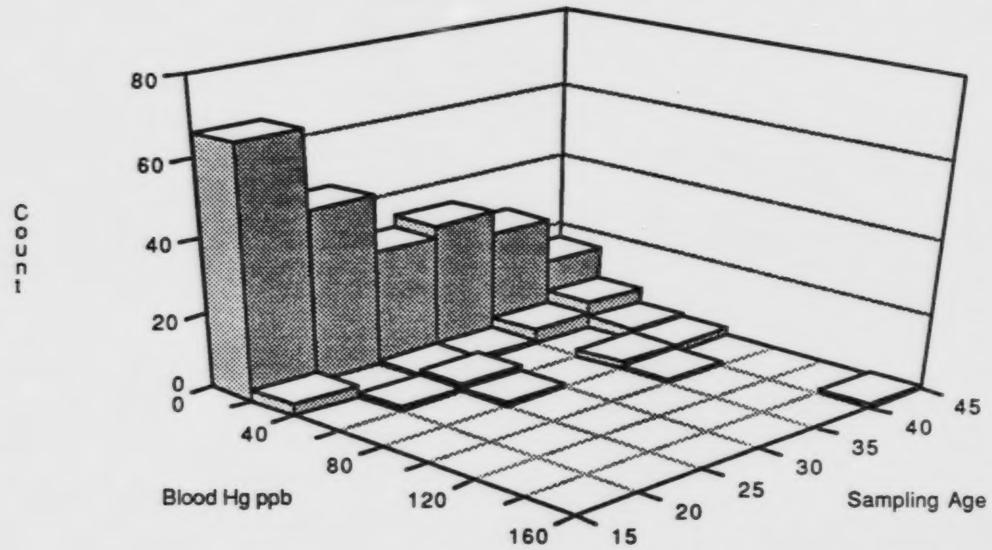
4.3.2.2.2 Blood levels



The distribution of blood mercury levels in all females sampled (N= 348). The highest reported value was 150ppb and the median value was 8ppb.



The distribution of mercury levels in females (N= 254) of child bearing age at the time of sampling. The highest reported level was 150ppb and the median value was 6ppb.



Nelson House

The distribution of samples according to the blood levels and the age at the time of sampling in females of child-bearing age. It may be seen that a number of samples exceed 20ppb in all age groups from 15 to 45 years. It is of concern that a number of samples are in the 40 to 100ppb range and that one sample has a value of 150ppb. There appears to be a tendency for the older women to have higher blood mercury levels.

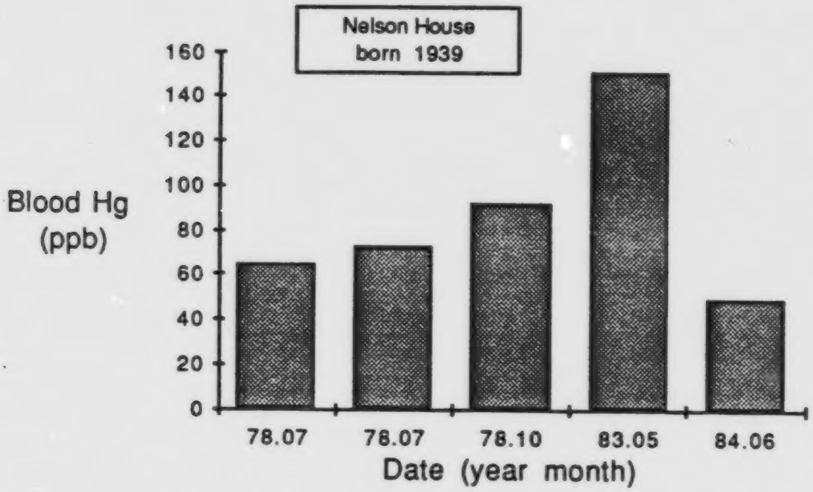
4.3.2.2.3 Cord blood levels

Birthdate		Blood Mercury
Year	Month	ppb
1982	June	2.4
1982	June	3.2
1983.	May	3.0

.The cord blood levels indicate no significant elevation of mercury levels. However, no conclusions can be drawn from these few cord blood values.

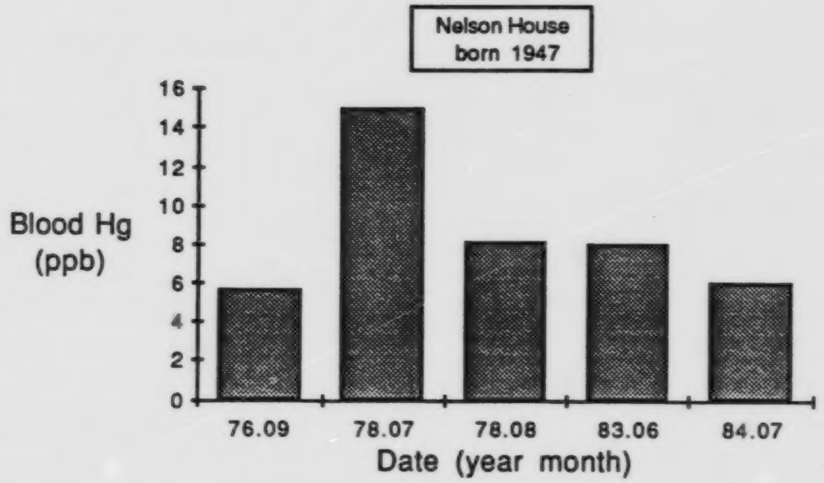
4.3.2.2.4 Repeat Samples

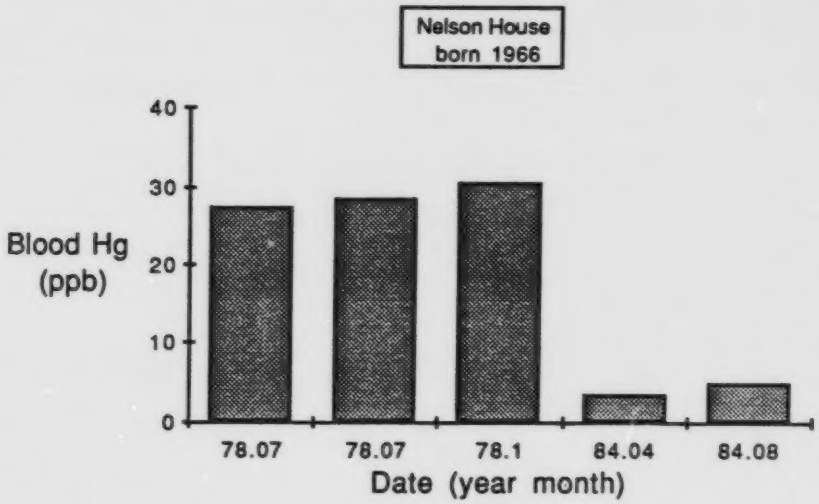
Below are presented five histograms indicating mercury levels in repeat samples from the same individuals. They are listed in order of the date of collection of the first sample.



The same date (78.07) are from two hair samples collected at different dates (January, 1979 and October 1978) but different segments were analysed so that the corresponding blood date was the same month (October, 1978). The agreement between the two values is within analytical error.

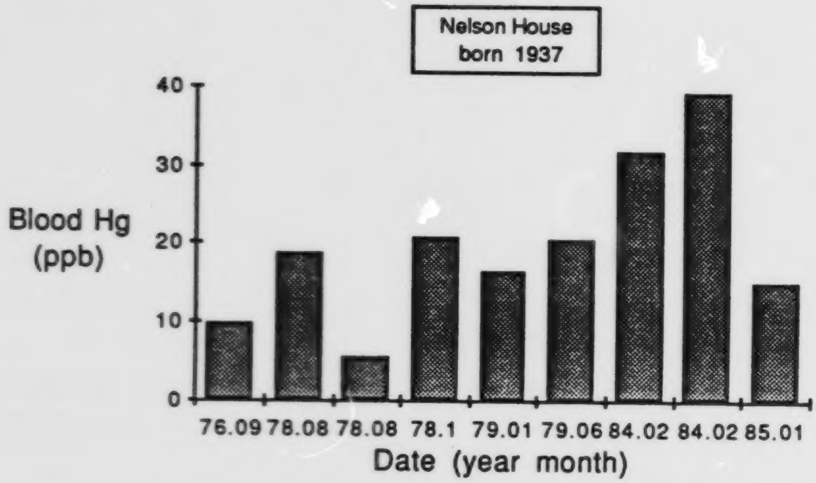
Nelson House



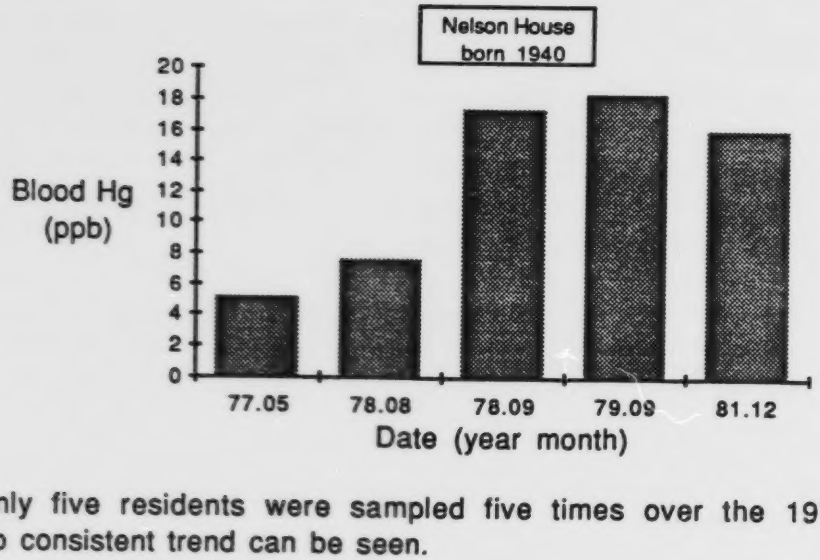


The first two samples have the same date of collection because of the position of the segment analysed. The first sample was actually collected in early 1979 but the segment analysed was 5 centimeters from the cut end. The second sample was collected in October 1978 but a segment two centimeters from the cut end was analysed. Thus the estimated date for the corresponding blood level was in the same month for both samples.

Nelson House



Two samples dated 78.08 (August, 1987) differ markedly. The lower value (5ppb) is from a blood sample collected at the beginning of the month whereas the higher value is calculated from the analysis of the first centimeter segment of a hair sample that would roughly correspond to the end of the month. Heavy fish intake during August could explain this difference.



Only five residents were sampled five times over the 1976-1985 period. No consistent trend can be seen.

4.3 INDIVIDUAL COMMUNITIES

4.3.3 YORK LANDING

SUMMARY AND EVALUATION

The distribution of mercury levels in this community indicates that the exposure to mercury is low with most values falling below the general accepted upper safe limit of 20ppb. However a number of samples of females of child-bearing age exceeded 20ppb to a maximum value of 60 ppb. No cord bloods were reported. No significant trend was seen in mercury levels according to date of collection.

The sampling results indicate that no significant health risks exist for adult or pre-natal poisoning. However, this conclusion must be somewhat qualified since no cord bloods were reported and a number of samples in females of child-bearing age exceeded 20ppb.

4.3.3.1 ALL SAMPLES

The table overleaf summarizes the data for all blood and hair samples collected over the sampling period from 1976 to 1986. It indicates the number of samples collected per year and the samples expressed as % of population. It also gives information on mercury levels. All mercury values have been converted to equivalent blood levels in ppb (parts per billion). Thus the number of samples are given falling in specified ranges of mercury levels. The latter are expressed as absolute numbers and as % of the samples collected. The lower part of the table gives a statistical estimate of the minimum and maximum number of samples expected to lie above the generally accepted health limit of 20ppb. The statistical procedures are described by Fleiss (1981) and estimate the 95% confidence limits for samples above 20ppb.

As may be seen from the table overleaf no sample ever exceeded 99ppb and an average of about 18% of the samples was in the range of 20 to 99ppb. A statistical analysis was unable to detect any downward trend in mercury values from 1976 to 1986.

Approximately 53% of the population were sampled at least once during the 10 year sampling period.

YORK LANDING (Summary of all samples)

Date	Samples number	0 -19ppb number	20 - 99ppb number	>99ppb number	Samples % population	0 - 19ppb % samples	20 - 99ppb % samples	>99ppb % samples
1976	17	9	8	0	4.0%	52.9%	47.1%	0.0%
1977	21	15	6		4.9%	71.4%	28.6%	0.0%
1979	73	56	17	0	17.2%	76.7%	23.3%	0.0%
1981	169	159	10	0	39.8%	94.1%	5.9%	0.0%
1982	6	3	3		1.4%	50.0%	50.0%	0.0%
1983	5	5	0	0	1.2%	100.0%	0.0%	0.0%
1984	41	26	15	0	9.6%	63.4%	36.6%	0.0%
1985	37	31	6	0	8.7%	83.8%	16.2%	0.0%
	Total	Total	Total	Total		Overall	Overall	Overall
Totals/Averages	369	304	65	0		82.4%	17.6%	0.0%

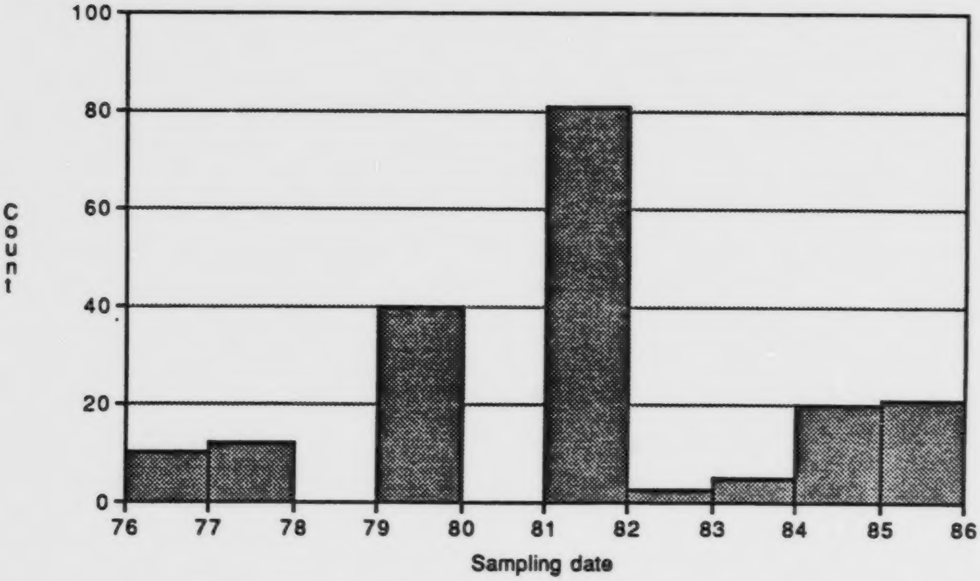
Date	Samples number	samples % population	> 20 ppb % samples	minimum % samples	maximum % samples
1976	17	4.0%	47.1%	23.9%	71.5%
1977	21	4.9%	28.6%	12.2%	52.4%
1979	73	17.2%	23.3%	14.5%	34.9%
1981	169	39.8%	5.9%	3.0%	10.9%
1982	6	1.4%	50.0%	13.9%	86.1%
1983	5	1.2%	0.0%		
1984	41	9.6%	36.6%	22.6%	53.1%
1985	37	8.7%	16.2%	6.7%	32.7%

The population or total number of residents in this community is assumed to be 425

4.3.3.2 ALL FEMALES

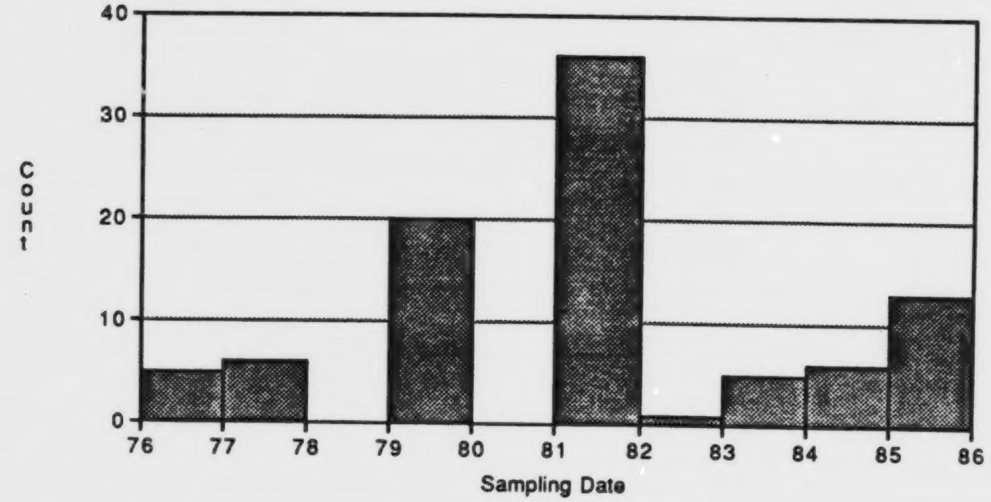
4.3.3.2.1 Sample collection

The histograms listed below summarize information on the number of samples collected in all females and also in females of child bearing age (assumed to be 15 to 45 years). Information is also summarized on the age at the time of sampling. However no correction has been made for the position of the segment on the hair sample so that both sampling date and sampling age may not correspond exactly with the actual date that the blood mercury concentration existed.

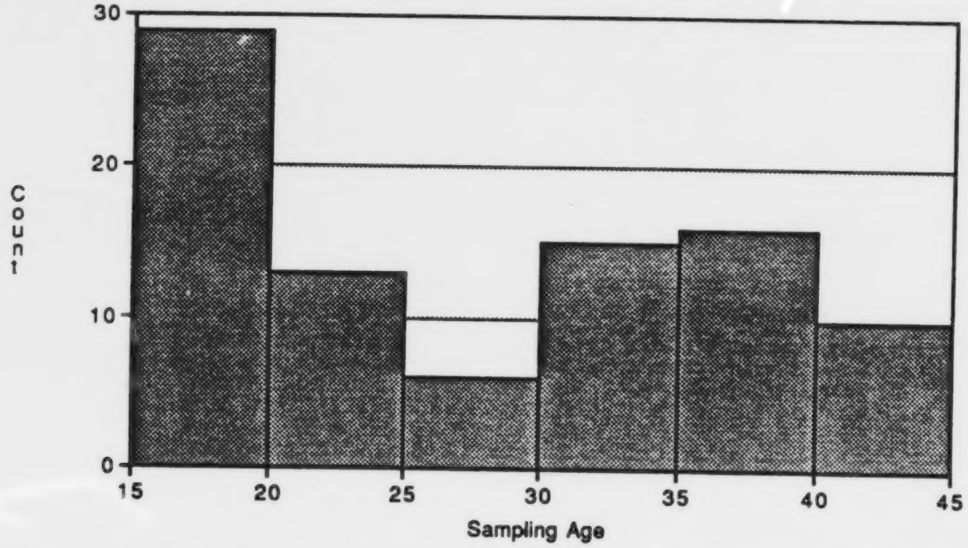


The number of samples collected according to the year of sampling. A total of 192 samples were reported.

YORK LANDING



The distribution of samples according to the date of sampling in all females of child bearing age (15 to 45 years). The distribution of samples was the same in females of child-bearing age as in all age groups.

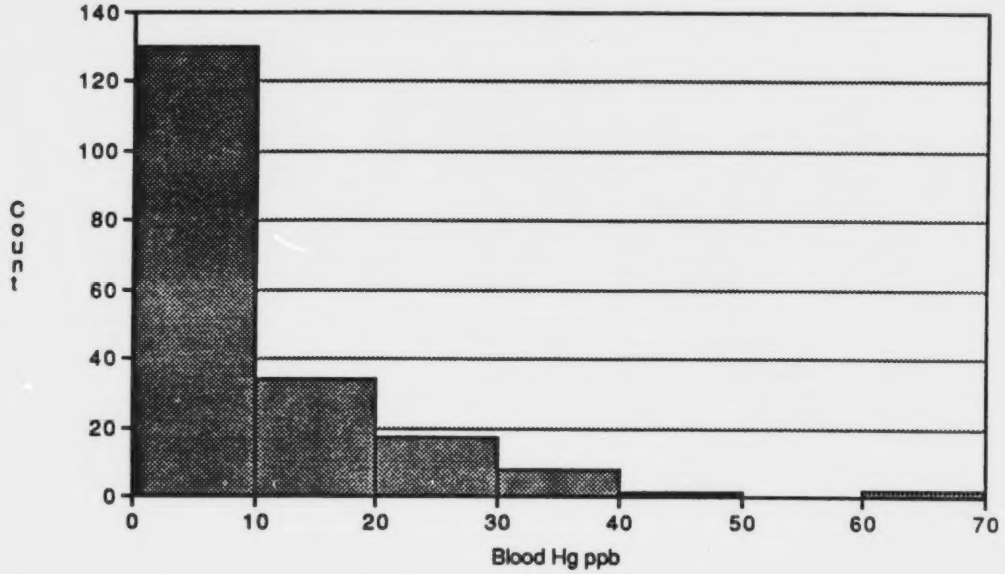


The distribution of samples according to age at the time of sampling in all females of child-bearing age (15 to 45 years). The histogram indicates that samples were collected from all age groups in the 15 to 45 year range

4.3.3.2.2 Blood levels

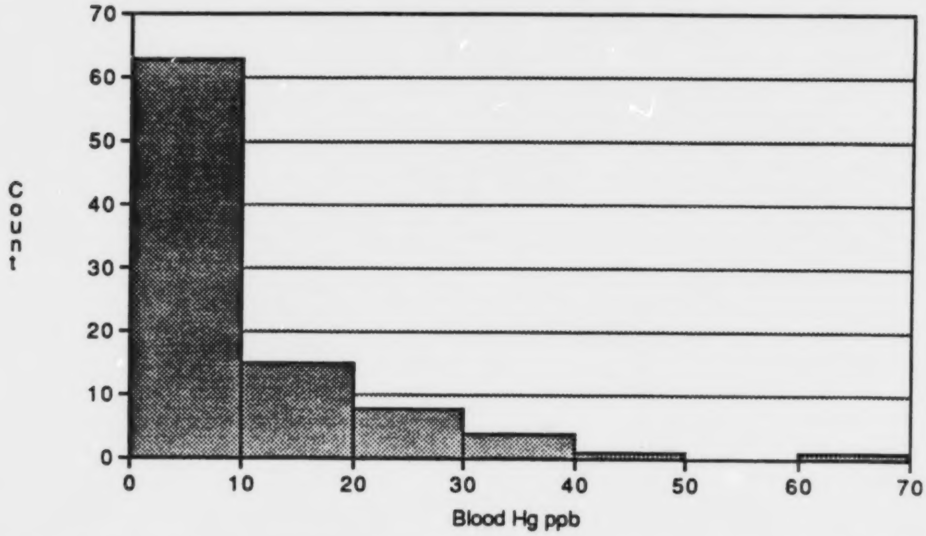
4.3.3.2.2 Blood levels

The two histograms below summarize information on blood levels of mercury.

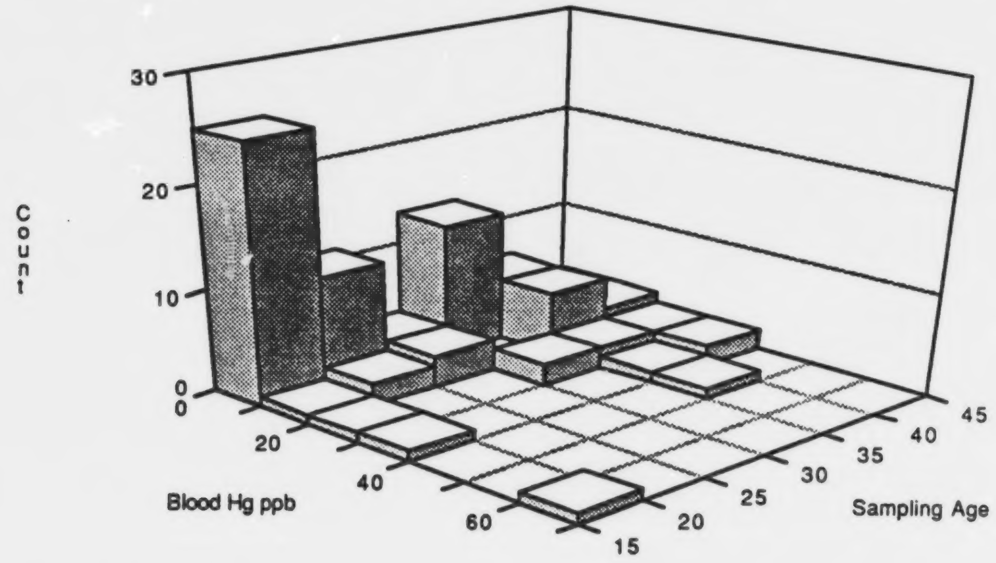


The distribution of blood mercury levels in all females. The highest reported value was 68 ppb and the median value was 6.9ppb.

YORK LANDING



The distribution of blood mercury levels in all females of child bearing age at the time of sampling. The highest value was 60ppb and the median was 6.9ppb. A number of samples exceeded 20ppb. In general the distribution of blood levels was the same in females of child-bearing age as compared to all age groups.



The distribution of samples according to the blood levels and the age at the time of sampling in females of child-bearing age. No trend of mercury levels with age is discernible.

4.3.3.2.3 Cord blood levels

No cord blood samples were reported.

4.3.3.2.4 Repeat Samples

Repeats of five samples or more were not reported.

4.3 INDIVIDUAL COMMUNITIES

4.3.4 SPLIT LAKE

SUMMARY AND EVALUATION

The distribution of mercury levels in this community indicates that the exposure to mercury is low with most values falling below the general accepted upper safe limit of 20ppb. However a number of females of child-bearing age had blood levels in excess of 20ppb extending to a maximum value of 45ppb. Only four cord bloods were recorded as compared to an estimated number of births of about 300 during the ten year sampling period. All cord bloods were below 12ppb. No significant trend was seen in mercury levels according to the year of collection.

The sampling results indicate that no significant health risks exist for adult or pre-natal poisoning. However, this conclusion must be somewhat qualified since few cord bloods were reported and a number of females of child-bearing age exceeded 20ppb.

4.3.4.1 ALL SAMPLES

The table overleaf summarizes the data for all blood and hair samples collected over the sampling period from 1976 to 1986. It indicates the number of samples collected per year and the samples expressed as % of population. It also gives information on mercury levels. All mercury values have been converted to equivalent blood levels in ppb (parts per billion). Thus the number of samples are given falling in specified ranges of mercury levels. The latter are expressed as absolute numbers and as % of the samples collected. The lower part of the table gives a statistical estimate of the minimum and maximum number of samples expected to lie above the generally accepted health limit of 20ppb. The statistical procedures are described by Fleiss (1981) and estimated the 95% confidence limits for samples above 20ppb.

As may be seen from the table overleaf no sample ever exceeded 99ppb and an average of about 17% of the samples was in the range of 20 to 99ppb. No significant trend is seen in these data with regard to year of sampling.Approximately 60% of the population were sampled at least once during the 10 year sampling period.

SPLIT LAKE (Summary of all samples)

Date	Samples number	0 -19ppb number	20 - 99ppb number	>99ppb number	Samples % population	0 - 19ppb % samples	20 - 99ppb % samples	>99ppb % samples
1976	31	25	6	0	3.0%	80.6%	19.4%	0.0%
1977	22	20	2	0	2.1%	90.9%	9.1%	0.0%
1979	170	145	25	0	16.4%	85.3%	14.7%	0.0%
1980	54	33	21	0	5.2%	61.1%	38.9%	0.0%
1981	240	204	36	0	23.1%	85.0%	15.0%	0.0%
1982	167	142	25	0	16.1%	85.0%	15.0%	0.0%
1983	9	9	0	0	0.9%	100.0%	0.0%	0.0%
1984	111	94	17	0	10.7%	84.7%	15.3%	0.0%
1985	66	53	13	0	6.4%	80.3%	19.7%	0.0%
	Total	Total	Total	Total		Overall	Overall	Overall
Total/overall	870	725	145	0		83.3%	16.7%	0.0%

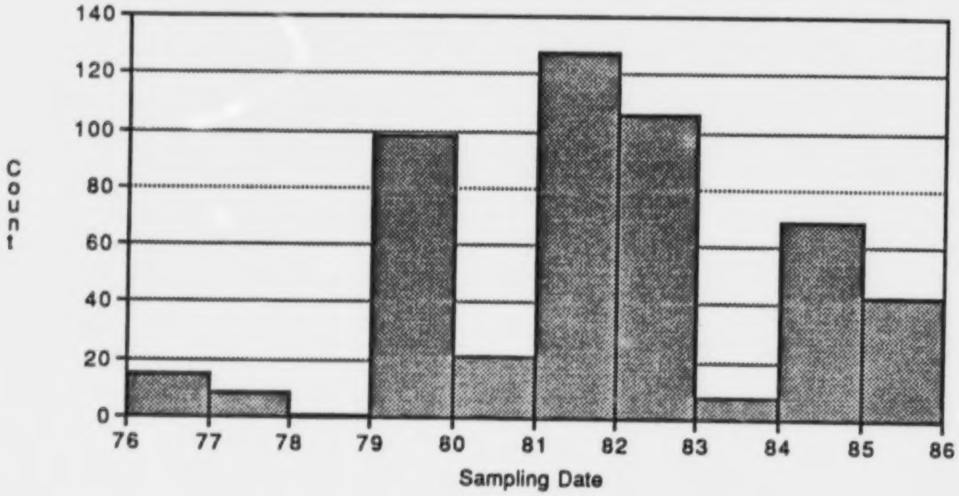
Date	Samples number	samples % population	> 20 ppb % samples	minimum % samples	maximum % samples
1976	31	3.0%	19.4%	8.2%	38.1%
1977	22	2.1%	9.1%	1.6%	30.6%
1979	170	16.4%	14.7%	9.9%	21.1%
1980	54	5.2%	38.9%	26.2%	53.1%
1981	240	23.1%	15.0%	10.9%	20.3%
1982	167	16.1%	15.0%	10.1%	21.5%
1983	9	0.9%	0.0%		
1984	111	10.7%	15.3%	9.4%	23.7%
1985	66	6.4%	19.7%	11.3%	31.7%

The population or total number of residents in this community is assumed to be 1037

4.3.4.2 ALL FEMALES

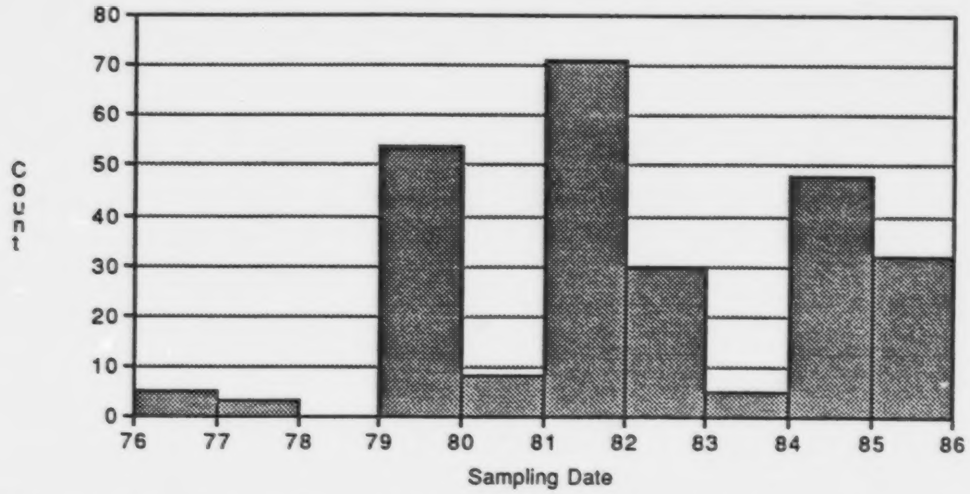
4.3.4.2.1 Sample collection

The histograms listed below summarize information on the number of samples collected in all females and also in females of child bearing age (assumed to be 15 to 45 years). Information is also summarized on the age at the time of sampling. However no correction has been made for the position of the segment on the hair sample so that both sampling date and sampling age may not correspond exactly with the actual date that the blood mercury concentration existed.

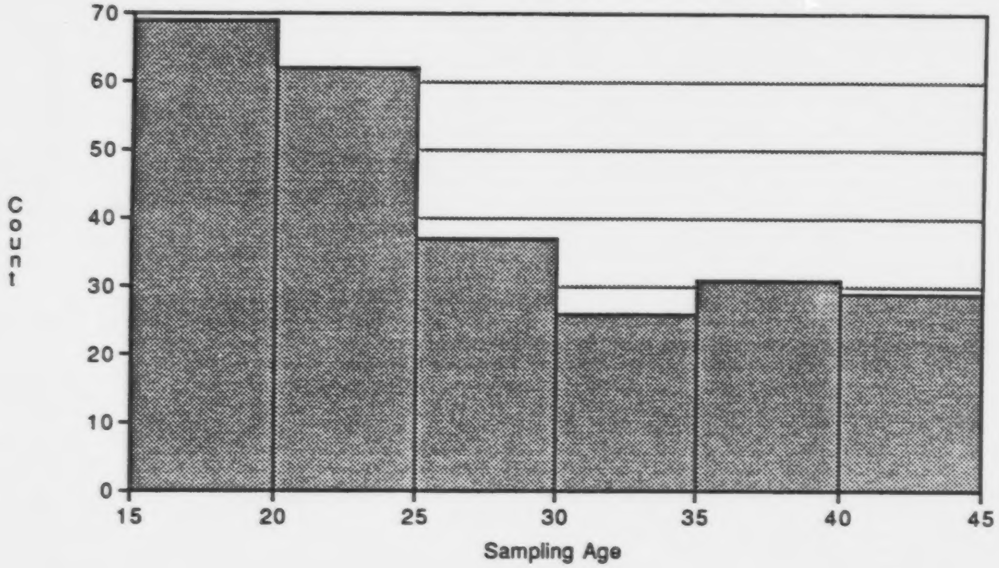


The number of samples collected from all females according to the year of sampling. A total of 479 samples were reported.

Split Lake



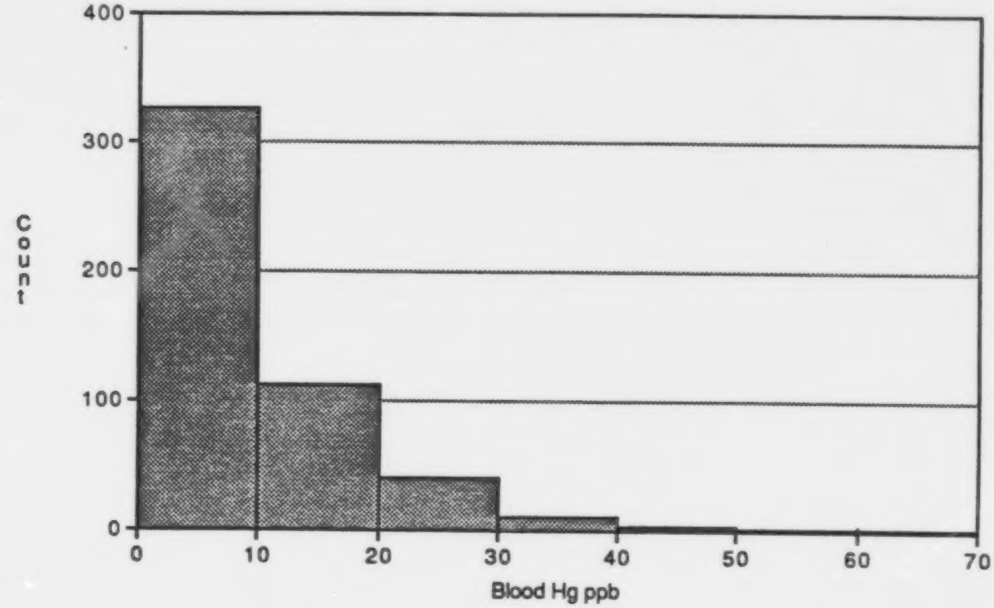
The number of samples collected according to the date of sampling in all females of child bearing age (15 to 45 years). The distribution of samples appears to be similar in females of child-bearing age as compared to all females.



The distribution of samples according to age at the time of sampling in all females of child bearing age (15 to 45 years). The histogram indicates that samples were collected from all ages in the age range from 15 to 45 years.

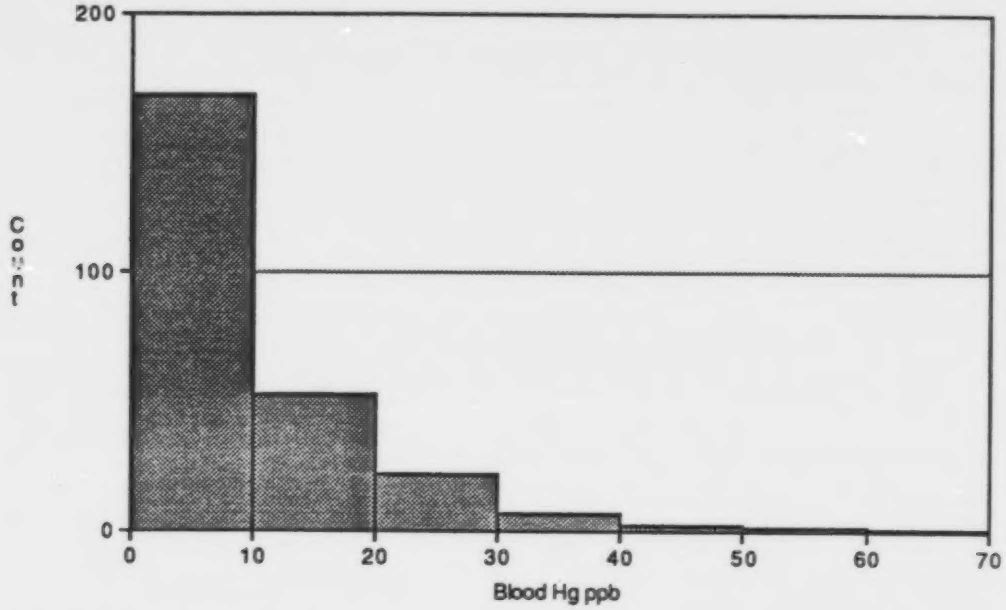
4.3.4.2.2 Blood levels

The two histograms below summarize information on blood levels of mercury in females.

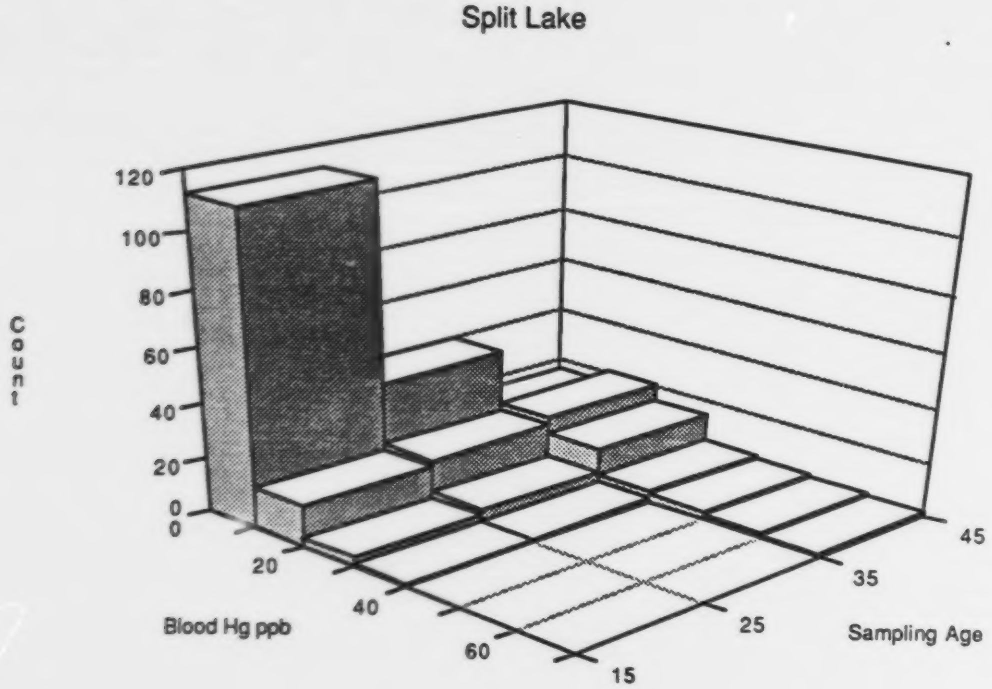


The distribution of blood mercury levels in all females sampled (N= 497). The highest reported value was 63ppb and the median value was 7.3ppb.

Split Lake



The distribution of blood mercury levels in all females of child bearing age at the time of sampling. The highest reported level was 63ppb and the median value was 7.0ppb. The distribution of mercury values in females of child-bearing age appears to be skewed to higher values as compared to all age groups. It may be seen that a number of samples exceed 20ppb.



The distribution of samples according to the blood levels and the age at the time of sampling in females of child bearing age. It may be seen that the higher values occur in the 35 to 45 year age group.

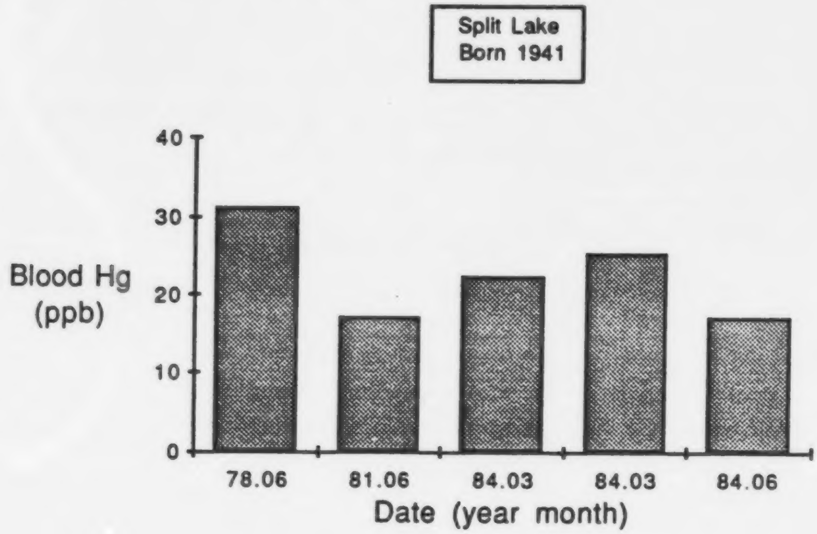
4.3.4.2.3 Cord blood levels

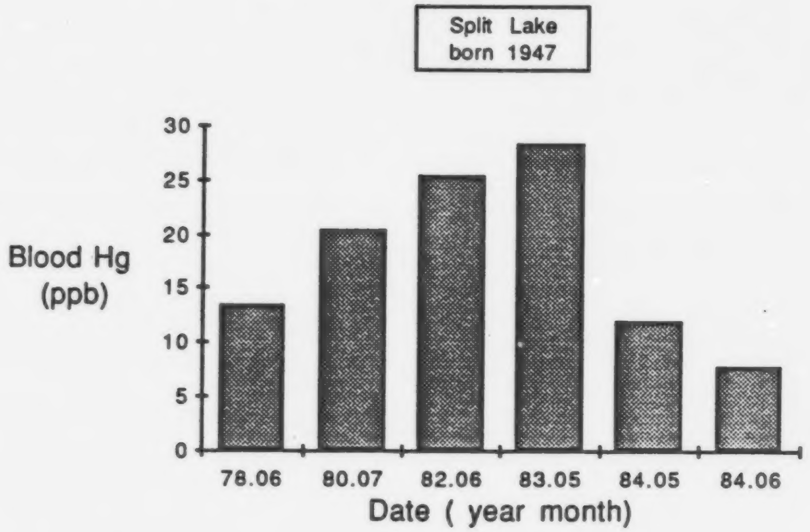
Birthdate		Blood Mercury
Year	Month	ppb
1983	February	7.20
1982	March	0.70
1983.	April	12.00
1983	June	2.20

The cord blood levels indicate no significant elevation of mercury levels.

4.3.4.2.4 Repeat Samples

Below are presented histograms indicating mercury levels in repeat samples from the same individuals.





Only two residents were sampled five times over the 1976-1985 period. No consistent trend can be seen.

4.3 INDIVIDUAL COMMUNITIES

4.3.5 CROSS LAKE

SUMMARY AND EVALUATION

The distribution of mercury levels in this community indicates that the exposure to mercury is low with most values falling below the general accepted upper safe limit of 20ppb. Females of child-bearing age are nearly all below 20ppb with the highest value being 22ppb. Only seven cord bloods were recorded as compared to an estimated number of births of about 500 during the ten year sampling period. All cord bloods were below 10ppb. A downward trend was seen in mercury levels in samples collected after 1979.

The sampling results indicate that no significant health risks exist for adult or pre-natal poisoning. However, this conclusion must be somewhat qualified since few cord bloods were reported and not more than 40% of the population were sampled over the ten year sampling period.

4.3.5.1 ALL SAMPLES

The table overleaf summarizes the data for all blood and hair samples collected over the sampling period from 1976 to 1986. It indicates the number of samples collected per year and the samples expressed as % of population. It also gives information on mercury levels. All mercury values have been converted to equivalent blood levels in ppb (parts per billion). Thus the number of samples are given falling in specified ranges of mercury levels. The latter are expressed as absolute numbers and as % of the samples collected. The lower part of the table gives a statistical estimate of the minimum and maximum number of samples expected to lie above the generally accepted health limit of 20ppb. The statistical procedures are described by Fleiss (1981) and estimate the 95% confidence limits on the percentage of samples above 20ppb.

As may be seen from the table overleaf, no sample ever exceeded 99 ppb and an average of about 8% of the samples was in the range of 20 to 99ppb. The % samples greater than 20ppb has a significant downward trend with time. However, there is no difference between years 1976 and 1979. Approximately 40% of the population were sampled at least once during the 10 year sampling period.

CROSS LAKE (Summary of all samples)

Date	Samples number	0 -19ppb number	20 - 99ppb number	>99ppb number	Samples % population	0 - 19ppb % samples	20 - 99ppb % samples	>99ppb % samples
1977	96	79	17	0	4.6%	82.3%	17.7%	0.0%
1978	1	1	0	0	0.0%	100.0%	0.0%	0.0%
1979	0	0	0	0	0.0%			
1980	70	58	12	0	3.4%	82.9%	17.1%	0.0%
1981	225	207	18	0	10.8%	92.0%	8.0%	0.0%
1982	296	275	21	0	14.2%	92.9%	7.1%	0.0%
1983	7	7	0	0	0.3%	100.0%	0.0%	0.0%
1984	200	192	8	0	9.6%	96.0%	4.0%	0.0%
1985	93	90	3	0	4.5%	96.8%	3.2%	0.0%
	Total	Total	Total	Total		Overall	Overall	Overall
Total/overall	988	909	79	0		92.0%	8.0%	0.0%

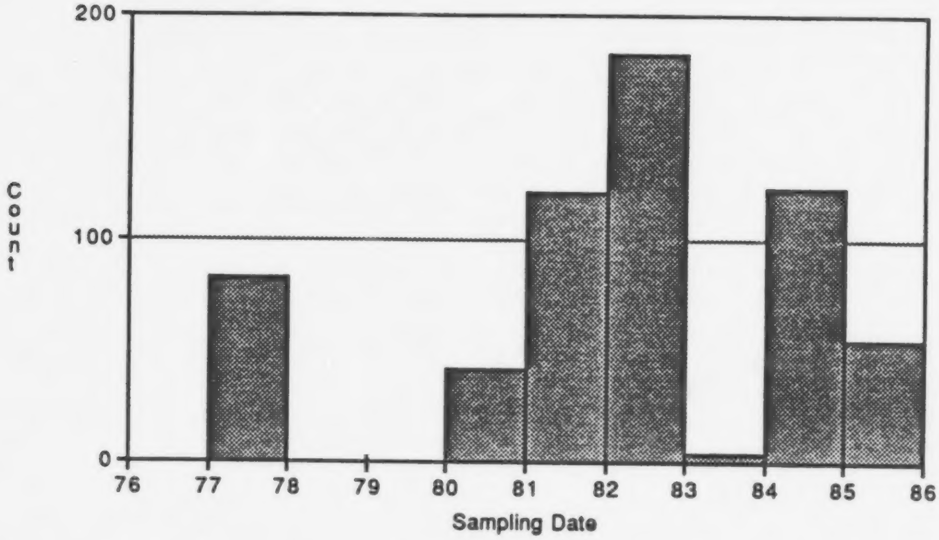
Date	Samples number	samples % population	> 20 ppb % samples	minimum % samples	maximum % samples
1977	96	4.6%	17.7%	10.9%	27.1%
1978	1	0.0%	0.0%		
1979	0	0.0%	0.0%		
1980	70	3.4%	17.1%	9.5%	28.4%
1981	225	10.8%	8.0%	4.9%	12.6%
1982	296	14.2%	7.1%	4.6%	10.8%
1983	7	0.3%	0.0%		
1984	200	9.6%	4.0%	1.9%	8.0%
1985	93	4.5%	3.2%	0.8%	9.8%

The population or total number of residents in this community is assumed to be 2097

4.3.5.2 ALL FEMALES

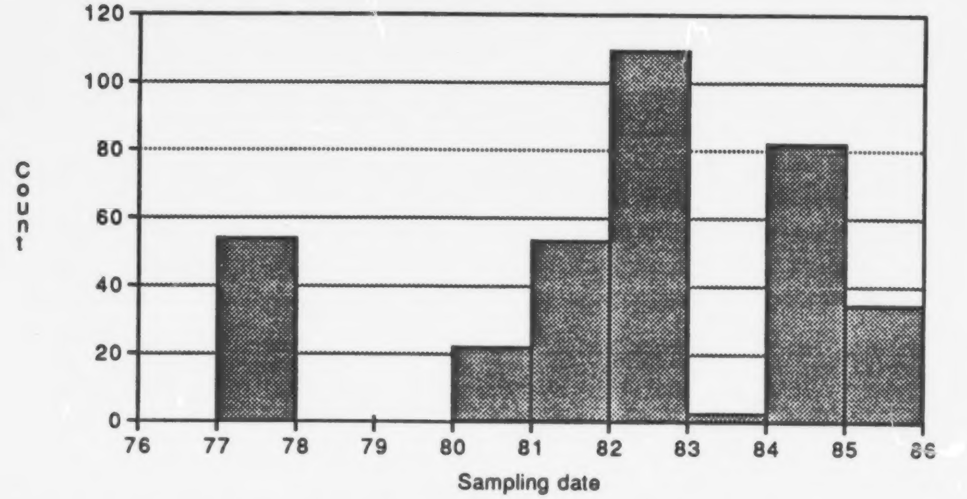
4.3.5.2.1 Sample collection

The histograms listed below summarize information on the number of samples collected in all females and also in females of child bearing age (assumed to be 15 to 45 years). Information is also summarized on the age at the time of sampling. However no correction has been made for the position of the segment on the hair sample so that both sampling date and sampling age may not correspond exactly with the actual date that the blood mercury concentration existed.

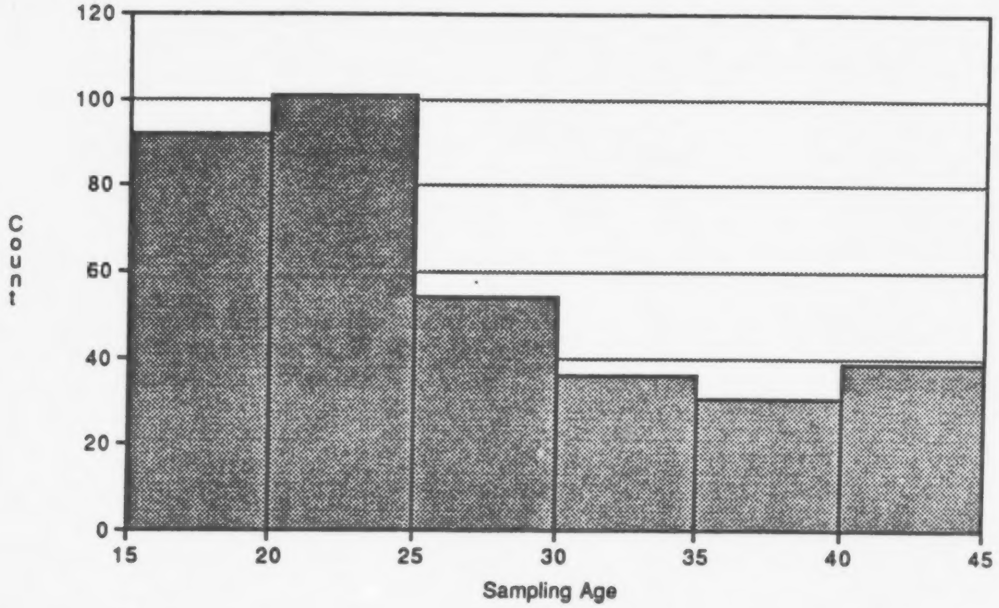


The number of samples collected according to the year of sampling. A total of 611 samples were reported.

Cross Lake



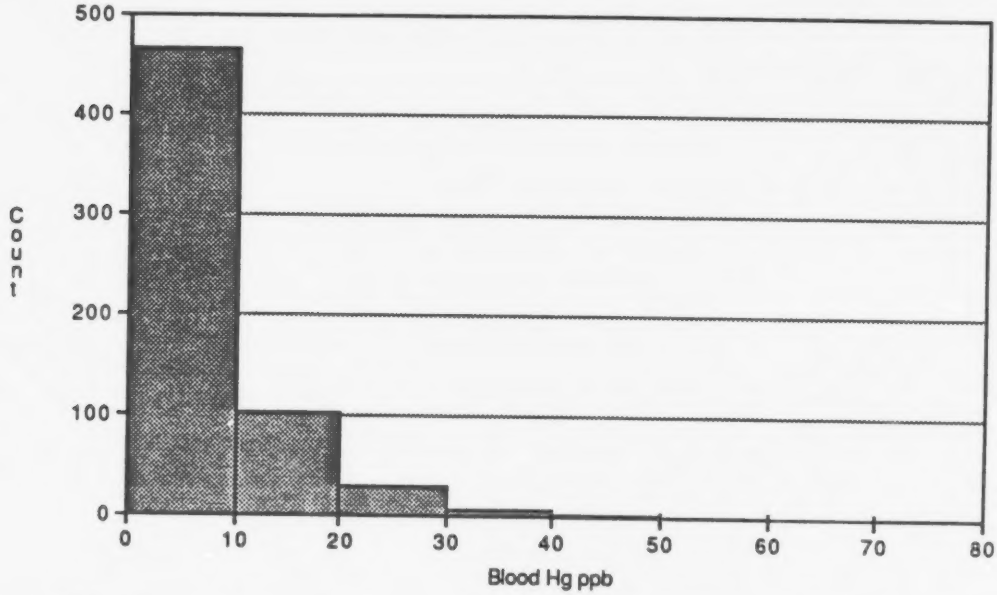
The number of samples collected according to the date of sampling in all females of child bearing age (15 to 45 years). The distribution of samples in females of child-bearing age appears to be the same as the distribution for all females sampled.



The distribution of samples according to age at the time of sampling in all females of child-bearing age (15 to 45 years). The histogram indicates that samples were collected from all age groups in the range from 15 to 45 years.

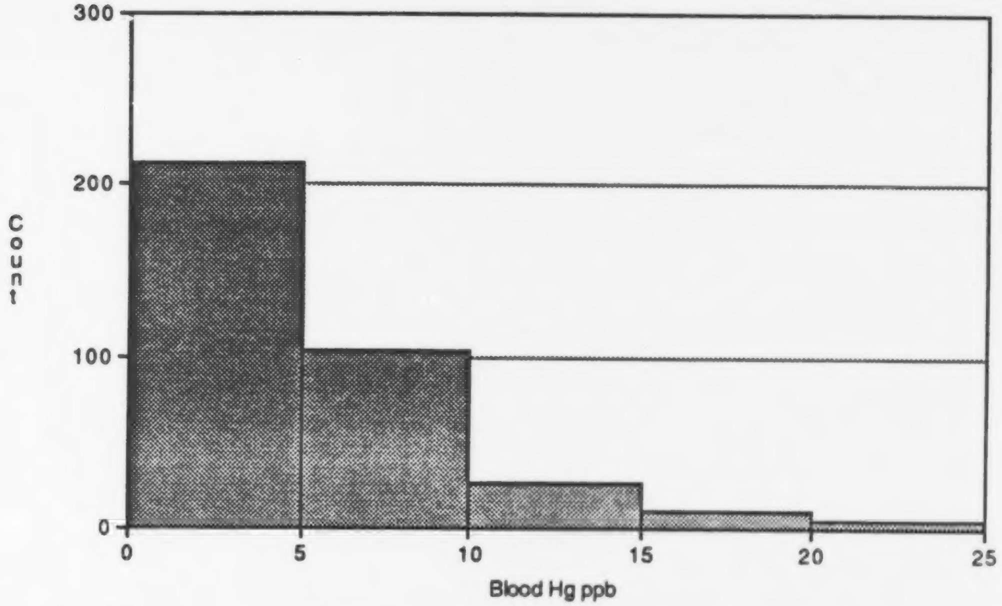
4.3.5.2.2 Blood levels

The two histograms below summarize information on blood levels of mercury in females sampled.



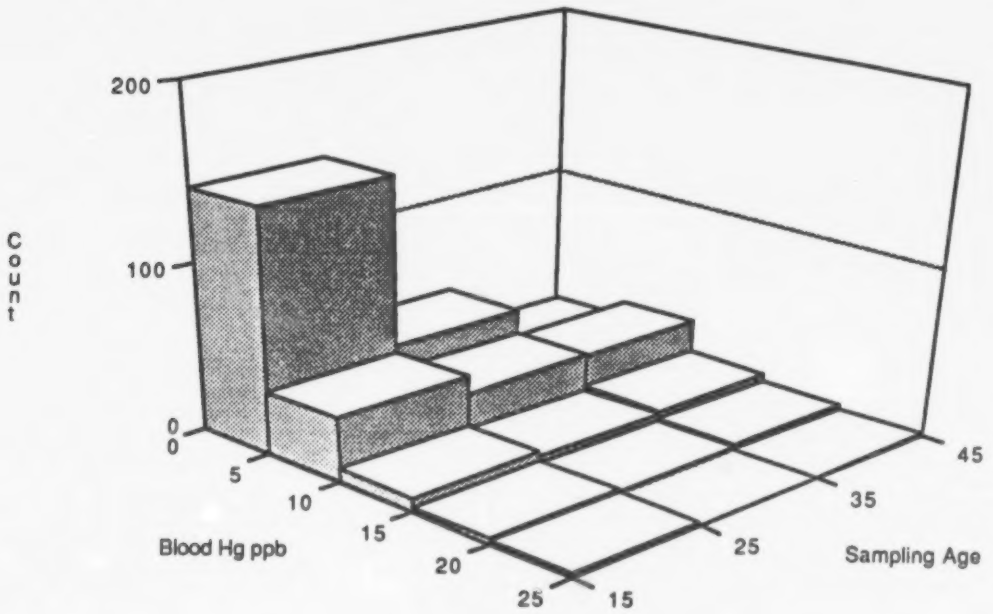
The distribution of blood mercury levels in all females sampled. The total number of observations was 611. The highest reported value was 77ppb and the median value was 5ppb.

Cross Lake



The distribution of blood mercury levels in all females of child bearing age at the time of sampling. The highest reported level was 22ppb. The distribution of blood levels in females of child-bearing age appears to be the same as in all females. Only a few samples exceed 20ppb and none exceed 25ppb.

Cross Lake



The distribution of samples according to the blood levels and the age at the time of sampling in females of child bearing age. The highest values were found in the 15 to 25 age group but this may reflect the fact that many more samples were collected in this group.

4.3.5.2.3 Cord blood levels

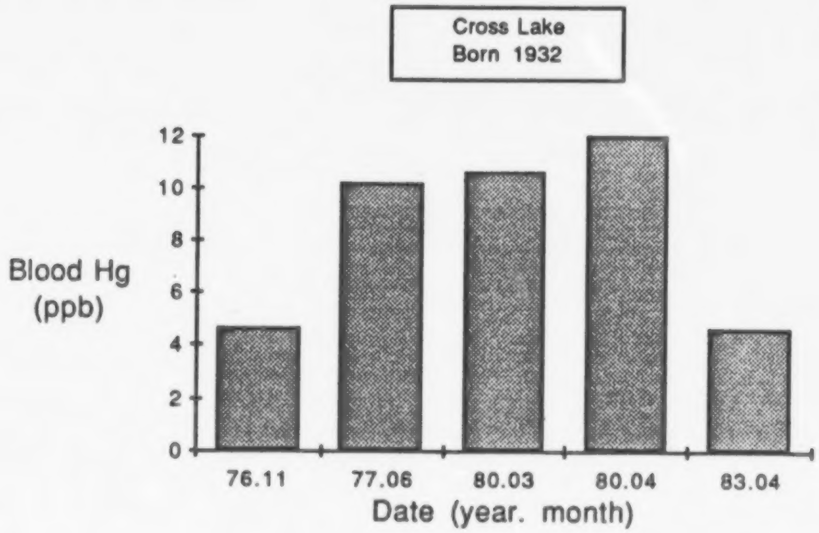
Birthdate Year	Month	Blood Mercury ppb
1982	May	7.7
1982	May	3.4
1982	August	6.3
1982	August	6.9
1983	February	7.1
1983	June	4.4
1983	April	6.1

The cord blood levels indicate no significant elevation of mercury levels.

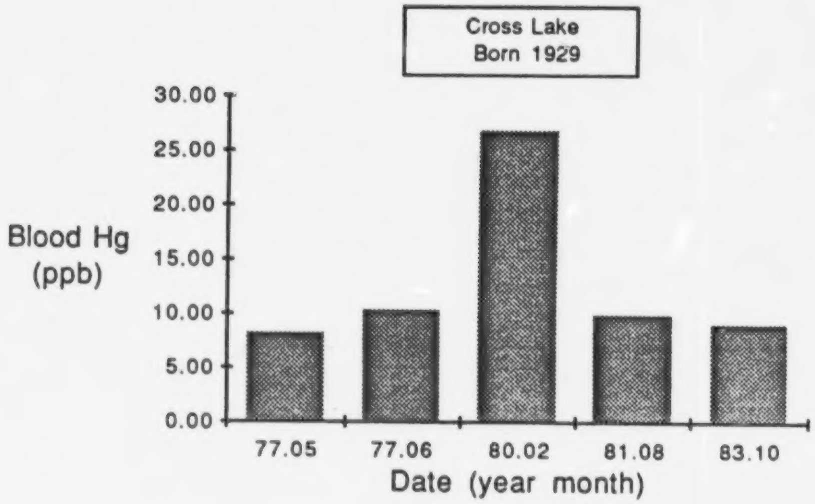
However, these limited data on cord bloods are consistent with the low mercury levels seen in samples from females of child bearing age.

4.3.5.2.4 Repeat Samples

Below are presented two histograms indicating mercury levels in repeat samples from the same individuals.



Cross Lake



Only two residents were sampled five times over the 1976-1985 period. No consistent trend can be seen.

4.3 INDIVIDUAL COMMUNITIES

4.3.6 NORWAY HOUSE

SUMMARY AND EVALUATION

The distribution of mercury levels in this community indicates that the exposure to mercury is low with most values falling below the general accepted upper safe limit of 20ppb. Females of child-bearing age are nearly all below 20ppb with the highest value being 42ppb. Forty three cord bloods were recorded from an estimated number of births of about 500 during the ten year sampling period. All cord bloods were below 10 ppb. No trend was seen in mercury levels in samples according to year of collection.

The sampling results indicate that no significant health risks exist for adult or pre-natal poisoning. However, this conclusion must be somewhat qualified since only 25% of the population were sampled over the ten year sampling period.

4.3.6.1 ALL SAMPLES

The table overleaf summarizes the data for all blood and hair samples collected over the sampling period from 1976 to 1986. It indicates the number of samples collected per year and the samples expressed as % of population. It also gives information on mercury levels. All mercury values have been converted to equivalent blood levels in ppb (parts per billion). Thus the number of samples are given falling in specified ranges of mercury levels. The latter are expressed as absolute numbers and as % of the samples collected. The lower part of the table gives a statistical estimate of the minimum and maximum number of samples expected to lie above the generally accepted health limit of 20ppb. The statistical procedures are described by Fleiss (1981) and estimate the 95% confidence limits for samples above 20 ppb.

No samples exceeded the 100ppb risk level for non-pregnant adults and an average of 6% of the samples was in the range of 20 to 99ppb. No significant trends were seen in mercury levels with date of sampling.

Approximately 25% of the population were sampled at least once during the 10 year sampling period.

NORWAY HOUSE (Summary of all samples)

Date	samples number	0 -19ppb number	20 - 99ppb number	>99ppb number	Samples % population	0 - 19ppb % samples	20 - 99ppb % samples	>99ppb % samples
1977	144	127	17	0	6.2%	88.2%	11.8%	0.0%
1978	81	75	6	0	3.5%	92.6%	7.4%	0.0%
1979	0	0	0	0	0.0%			
1980	66	59	7	0	2.8%	89.4%	10.6%	0.0%
1981	107	99	8	0	4.6%	92.5%	7.5%	0.0%
1982	127	126	1	0	5.4%	99.2%	0.8%	0.0%
1983	54	54	0	0	2.3%	100.0%	0.0%	0.0%
1984	221	212	9	0	9.5%	95.9%	4.1%	0.0%
	Total	Total	Total	Total		Overall	Overall	Overall
Total/overall	800	752	48	0		94.0%	6.0%	0.0%

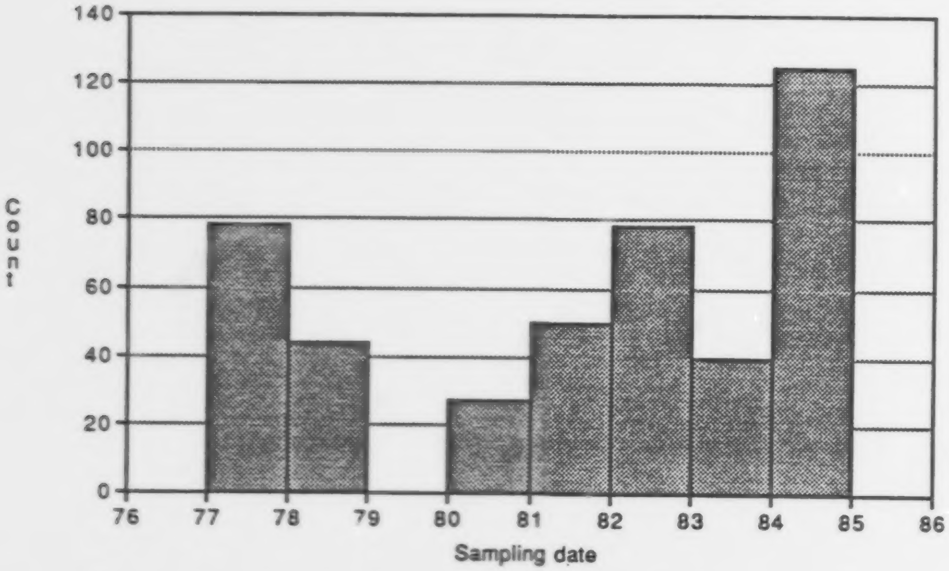
Date	Samples number	samples % population	> 20 ppb % samples	minimum %samples	maximum %samples
1977	144	6.2%	11.8%	7.2%	18.5%
1978	81	3.5%	7.4%	3.0%	16.0%
1979	0	0.0%			
1980	66	2.8%	10.6%	4.7%	21.2%
1981	107	4.6%	7.5%	3.5%	14.7%
1982	127	5.4%	0.8%	0.0%	5.0%
1983	54	2.3%	0.0%		
1984	221	9.5%	4.1%	2.0%	7.9%

The population or total number of residents in this community is assumed to be 2335

4.3.6.2 ALL FEMALES

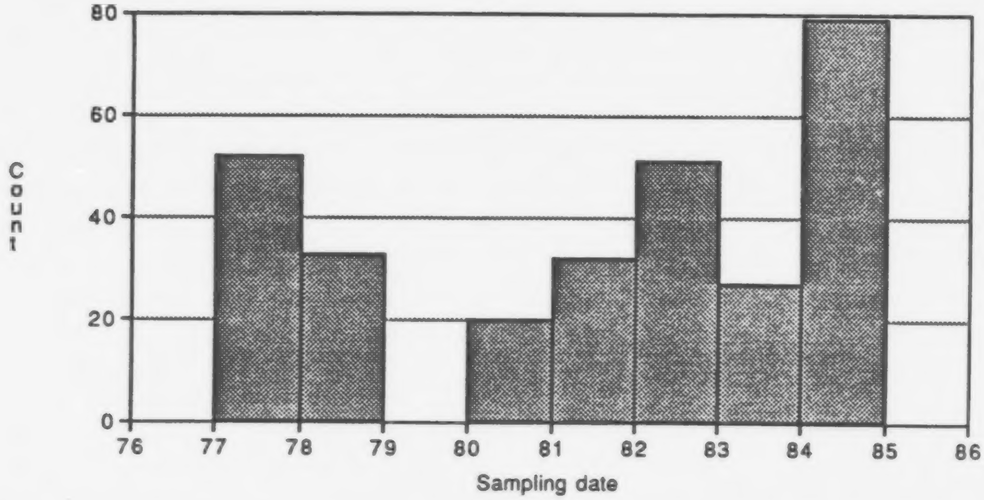
4.3.6.2.1 Sample collection

The histograms listed below summarize information on the number of samples collected in all females and also in females of child bearing age (assumed to be 15 to 45 years). Information is also summarized on the age at the time of sampling. However no correction has been made for the position of the segment on the hair sample so that both sampling date and sampling age may not correspond exactly with the actual date that the blood mercury concentration existed.

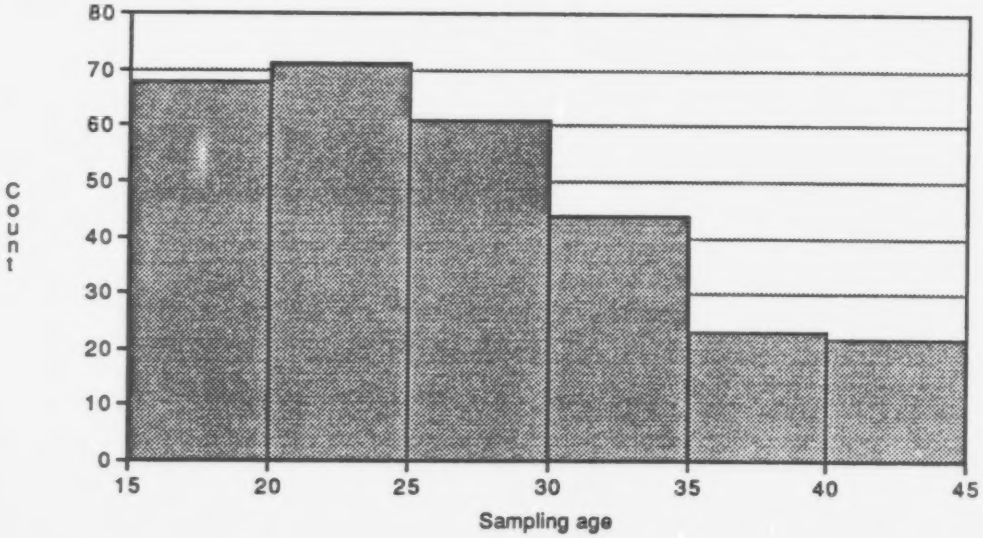


The number of samples collected according to the year of sampling. A total of 443 samples were reported.

NORWAY HOUSE



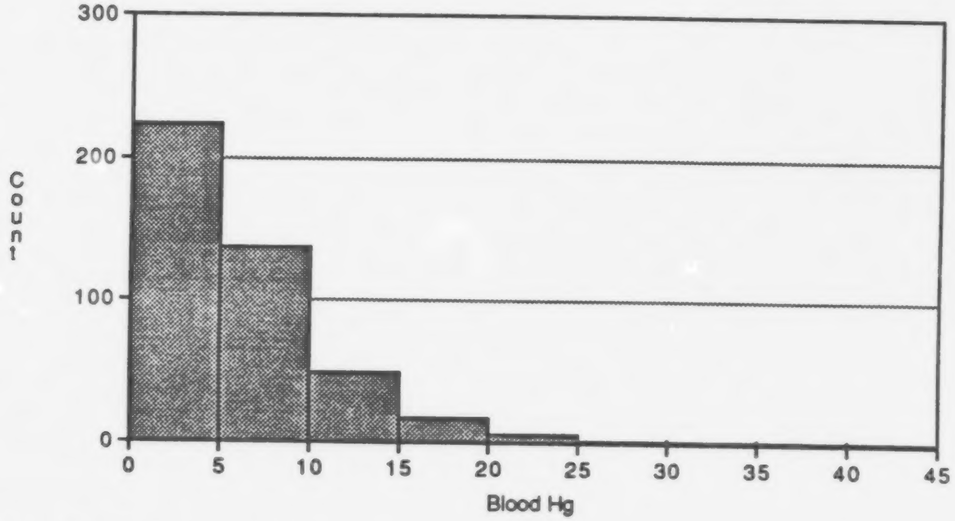
The distribution of samples according to the date of sampling in all females of child-bearing age (15 to 45 years of age). The distribution in females of child-bearing age appears to be the same as for all females sampled.



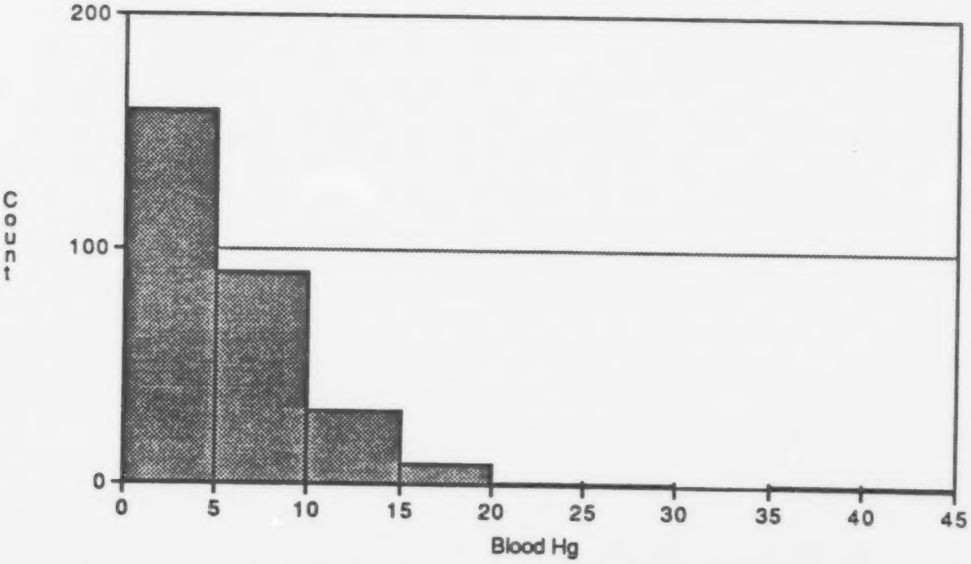
The distribution of samples according to age at the time of sampling in females of child-bearing age. The histogram indicates that samples were collected from all ages in the range from 15 to 45 years.

4.3.6.2.2 Blood levels

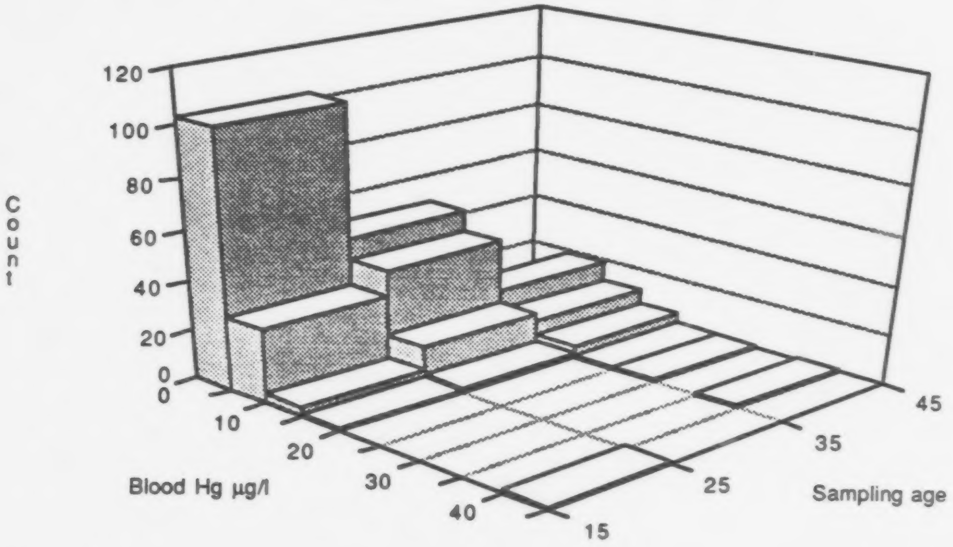
The two histograms below summarize information on blood levels of mercury.



The distribution of blood mercury levels in all females sampled. The total number of observations was 443. The highest reported value was 42ppb and the median value was 5ppb.



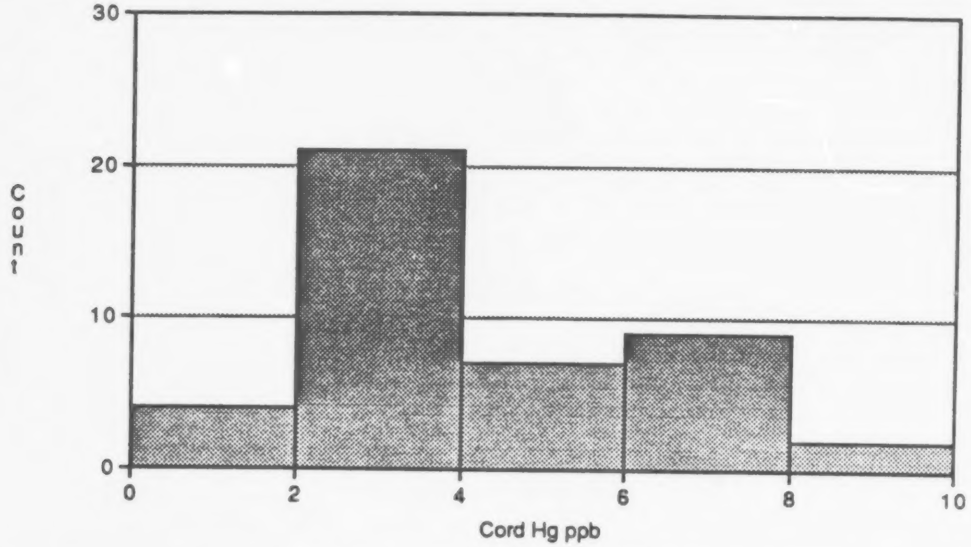
The distribution of blood mercury levels in all females aged 15 to 45 years at the time of sampling. The highest reported level was 42ppb and the median value was 4.3ppb. The distribution of blood levels in females appears to be the same as that seen in all females. Only a few samples exceeded 20ppb.



The number of samples according to the blood mercury levels and the age at the time of sampling in females of child-bearing age. The higher values tended to be in the 35 to 45 year age group except for the highest sample which was in the 15 to 25 year age group.

NORWAY HOUSE

4.3.6.2.3 Cord Blood levels



The histogram indicates all mercury levels fall in a narrow range and all are below 10ppb.

Data File: Norway cord
Variable: Cord Hg ppb Observations: 43

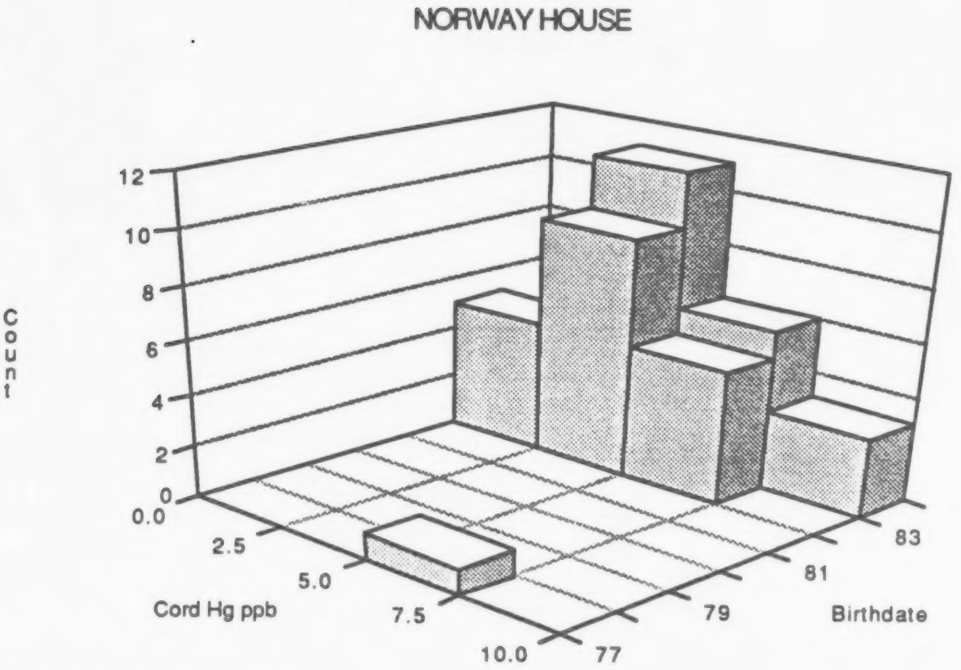
Minimum: 1.500 Maximum: 9.400
Range: 7.900 Median: 3.900

Mean: 4.326 Standard Error: 0.306

Variance: 4.021
Standard Deviation: 2.005
Coefficient of Variation: 46.355

Skewness: 0.666 Kurtosis: -0.347

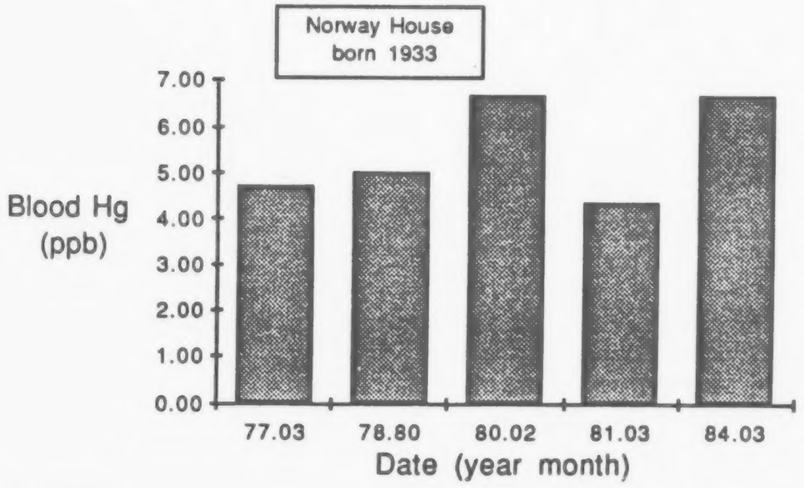
The descriptive statistics indicate the remarkably tight grouping of the data and the very low average concentration of mercury.



The 3D-histogram indicates that virtually all the samples were collected over a two year period (1982-1983) so that it is impossible to detect any trends with time.

4.3.6.2.4 Repeat Samples

Only one repeat of five samples or more was reported for this community.



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APPENDIX I

DETAILS OF MERCURY SAMPLING PROGRAM

The mercury sampling program is detailed in five sections:

1. Summary
2. Target Groups
3. Sample Collection
4. Laboratory Analytical Procedures
5. Reporting Results and Follow-up

1. SUMMARY

The following steps are taken to initiative the mercury sampling program in a community:

(a) Organize a meeting with the Chief and Council.

(b) Meet the local health committee, if any.

(c) Community Health Nurses, Community Health Representatives and Indian organization mercury workers, if any, should be involved in these meetings.

(d) Give a full explanation of the program with reasons for sampling and planned follow-up.

(e) Before sampling can begin, permission must be obtained from the Band Council.

(f) In most communities, it should be possible to hold public meetings to outline objectives and methods of the survey.

(g) Explain the entire procedure to each family or individual involved.

(h) When test results become available the following procedure is to be followed:

   (i) Laboratory sends results to Regional and Branch Headquarters.
   (ii) Branch sends a summary to Region.
   (iii) Region sends the summary plus copies of individual's results to the nurse-in-charge.
   (iv) Region sends a summary of each community's results to the Band Council. Such a summary will list the number of results in each of the categories listed above under the heading, "Medical Services Branch Standards". It will not contain any names or other personal, identifying information.
   (v) Region sends individual results to the nurse-in-charge of the appropriate nursing or health centre.

(vi) With the assistance of CHR's and, where available, mercury workers, the CHR's arrange for visits to the households for which results are available so that each family may be given the sample results for the individual members. These should be in writing and should be accompanied by a verbal explanation as follows:

Result normal - no further explanation required.
Result in increasing risk category - individual told that mercury level elevated but still within acceptable limits.
Result in at risk category - individual told that mercury level is high and arrangements are made for them to be further tested and advised about eating fish.

2. TARGET GROUPS

The following Target Groups have been shown to be the most likely to have high levels of mercury in their bodies (a,b) or to be at special risk (d,e). Heads of Households (c) are included since it has been found that the level of mercury in the head of the household usually is indicative of the mercury levels in other members of the household (Clarkson, 1976).

(a) "Heavy" fish eaters
(b) Fishing guides and their families
(c) Heads of households
(d) Small children
(e) Pregnant women

A hair sample is being requested as part of the standard investigation in the antenatal care of women from communities in areas where the community profiles are in the higher range. A hair sample will indicate the pregnant woman's pattern of exposure to mercury over the preceding year and help forecast the likelihood of her reaching significant levels during her pregnancy.

A pregnant woman found to have a blood mercury level above 20 ppb is provided with suitable advice regarding modification of her diet. If the mercury level approaches or exceeds 100 ppb, the advice must be very intensive and all possible efforts used to lower the mercury level without compromising the pregnant woman's nutritional status.

Medical Services Branch developed a pamphlet entitled, "Information about Mercury" April 1981, (Appendix I) which provides information about mercury, the significance with advice as to what people that may have high mercury levels should do, i.e. eat less fish.

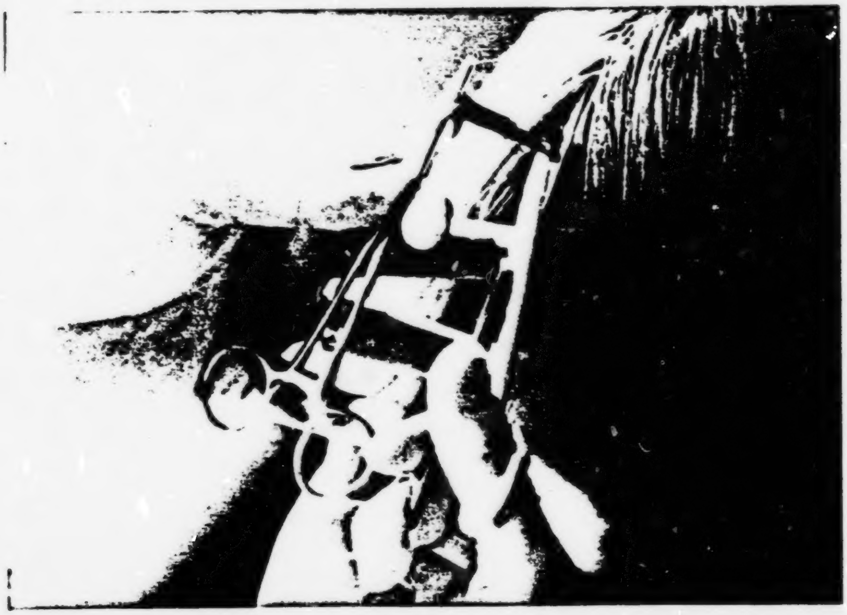
Persons who are considered "at risk" are entered on the Chronic Condition register, and listed as having high level of mercury in their blood. These persons are checked by the nurse/doctor on a regular basis along with additional tests.

There is on-going monitoring of those communities in which it is known that there is a high mercury content. Also, health education is provided by the health care workers, i.e. Nurse, Community Health Representative, etc. Phamplets are issued and posters are displayed in Nursing Stations and in various locations within the community.

3. SAMPLE COLLECTION

Since hair grows at an approximate rate of one centimetre per month and since the amount of mercury in each centimetre reflects the amount of mercury ingested during the period of growth of that centimetre, it follows that a well-taken hair sample will provide one with a record of the donor's exposure to contamination for as many months past as the hair is centimetres long.

Before any sampling can begin, permission must be obtained from the Band Council. The Medical Services Branch nurse and/or the Community Health Representative collect the hair and blood samples during the performance of their regular duties. Sampling is concentrated on heads of households, heavy fish eaters, fishing guides, pregnant women, women of child-bearing age, and those having had "at risk" levels. Usually the CHR's (community member qualified by virtue of completion of the training course who assist the Indian people on reserve to reach and maintain the highest possible standard of health) can identify heavy fish-eaters through their knowledge of the individual and the community. Verbal consent is accepted as an agreement to take a sample from an individual; however, written consent is generally obtained for children under four years old. An explanation of the objectives and procedures of the mercury testing program is given to all individuals, community leaders, and families before consent is obtained and sampling is performed.



To take a hair sample, a lock about the size of regular pencil width is selected from the area where the hair is longest. Pincers are used to select and hold the hair while it is cut close to the scalp. The hair sample is then placed in a plastic bag and stapled two or three times to keep it in place. It is important to keep the cut ends of the hair even. A line is marked on the bag at the proximal end of the sample. The individual's name, birthdate, band name and number are also written on the bag. An Environmental Contaminant Sample Report form is filled out by the sampler, which gives identifying information on each individual, type of sample and what it is to be analyzed for. This completed form is stapled to the outside of the plastic bag.

After several hair samples have been taken, they are packaged together and sent to the Medical Services Branch Occupational Health Unit Laboratory in Ottawa.

A blood specimen is collected in a mercury-free Heparinized tube and labelled with the individual's name, birthdate, band name and number. An Environmental Contaminant Sample Report form is filled out in the same manner as with a hair sample. The specimens were placed in ice or ice packs in a styrofoam cooler, as soon as possible after the sampling. Refrigerated Heparinized whole blood is stable for approximately two weeks and, therefore, samples were sent immediately to the Occupational Health Unit Laboratory in Ottawa for analysis. Care should be taken to ensure that the blood specimens are not frozen, as this can cause an interference in the analytical determination.

Arrangements are made with hospitals to have maternal and cord blood samples taken at the time of delivery and shipped in the same manner to the Ottawa Laboratory.

4. LABORATORY ANALYTICAL PROCEDURES

Once the samples reach the Occupational Health Unit Laboratory, they are assigned a file number. Each file contains a record of the pertinent information regarding the group of samples. This information includes: type of sample, analysis required, location where sample originated, related files in the case of multiple determinations, information necessary for measuring the Laboratory's analytical output and information necessary for indicating the calibration and quality control analyses which are not counted towards the analytical output.

There is a second numbering system used for hair samples, due to the large number of sub-samples generated by cutting the hair for segmental determinations. This number is for internal use only and specifies the region, a sample number within that region and the distance from the scalp. Only the first 12 centimetres of a hair sample are analysed, as this gives a picture of the mercury intake over the last year.

An adaptation of the Magos Method is used for the determination and speciation of mercury in blood, hair, tissue and urine specimens. This method is based on the cold vapor atomic absorption spectrophotometric detection of elemental mercury. These analyses involve complexing the mercury compounds in the sample with L-cysteine at high pH. Subsequent addition of a reducing agent reduces mercury present to an elemental vapour which is swept through a fixed wavelength analytical cell. A linear detector response is obtained over several orders of magnitude. The relative strength of the reducing agent selected determines which of the mercury compounds are reduced to the elemental vapor. Stannous chloride reduces only ionic species present. This measurement reveals the inorganic portion of the mercury present in the sample. If a combination of stannous chloride and cadmium chloride is added to the sample, covalently bound mercury compounds are also reduced. This allows determination of the total mercury concentration. Organic mercury concentrations are calculated from the difference between the total mercury and inorganic mercury concentrations (Magos, 1971); Farant et al, 1981).

All results are recorded on the individual's Environmental Contaminant Samples Report form, however the highest level is the recorded value. A covering letter signed by the Director of the Occupational Health Unit accompanies a group of similar E.C.S.R. forms which indicates the report number, file number, date samples were received in the laboratory and where the samples were collected. Copies of this package go to the Region, the Director of Environmental Health Services, and the Occupational Health Unit retains one. All reports are considered "Medical Confidential" and appropriate measures are taken to ensure the confidentiality of the individual.

The turnaround time from receipt of the samples in the Laboratory to the reporting date is determined by the number of samples arriving at one time, the backlog of samples for mercury determinations and the availability of analysis. Generally, for blood specimens it is about two weeks or less, and for hair samples it is about four weeks or less.

5. REPORTING RESULTS AND FOLLOW-UP

When the Manitoba Regional Office of Medical Services Branch receives the results, a summary of the community's levels is sent to the Chief and Council. This summary indicates how many samples were taken and which of the following ranges they fall into: 0-19 ppb, 20-99 ppb, and greater than 100 ppb. This summary does not contain any names or other personal identifying data.

A list of individual results, with names, date of birth, band name and number is sent to the nurse-in-charge of the appropriate nursing station or health centre. With asistance of the CHRS, where possible visits are made to the households so that results can be given to the individuals along with a verbal explanation. Also a form letter, signed by the Regional or Zone

Director, is sent to the individual who was sampled indicating the date the sample was taken and what the result was. A sheet explaining what the various levels of mercury mean is attached for information. In the case of elevated levels in a woman of child-bearing age, it is recommended that this individual see the nurse-in-charge for an explanation of the significance of the level and what follow-up procedures may be necessary. For a man, with levels betweeen 20-99, another sample is recommended at a later date. If individuals have a result greater than 100 ppb or a pregnant woman has a level greater than 20 ppb, then the nurse-in-charge would request that they have a repeat blood and hair sample taken as soon as possible. They are also counselled as to appropriate actions to lower their levels.

A neurological examination is performed if the individual's mercury level is greater than 200 ppb. The Regional Office determines when an examination should be offered and arrangements are made through the nurse-in-charge. Repeat hair samples are sought annually from all individuals with mercury values greater than 20 ppb, until such time as their mercury profiles fall within the normal guideline.

Communities and individuals are being provided with information on mercury levels in local fish species, since a careful balance must be established between the advantage of the traditional Indian and Inuit way of life, which can provide an excellent source of dietary protein in the form of fish and sea mammals, and the potential danger to health due to elevated mercury levels in that food source.

Fish consumption guidelines suggest maximum permissible mercury level in fish for the Canadian commercial fishery is 0.5 ppm.

The maximum suggested for fish eaten by Indian people in Canada is less than 0.2 ppm, unless fish is an insignificant part of the diet.

Safe Weekly consumption levels of fish are shown as Table 1:

TABLE 1

SAFE WEEKLY CONSUMPTION LEVELS OF FISH

Mercury Level in Fish	Safe Fish Consumption Per Week
1.0 ppm	0.34 lb. (0.16 kg)
0.5 ppm	0.68 lb. (0.31 kg)
0.4 ppm	0.86 lb. (0.39 kg)
0.3 ppm	1.14 lb. (0.52 kg)
0.2 ppm	1.71 lb. (0.78 kg)
0.1 ppm	3.42 lb. (1.56 kg)

The safe fish consumption per week is estimated for a 70 kg. man assuming 20 ppb to be the maximum acceptable human blood level equivalent to a maximum weekly intake of approximately 0.15 mg Mercury as Methylmercury. These safe consumption levels are intended to protect the fetus. A person of more or less weight could consume respectively more or less fish in proportion to their weight as compared to a 70 kg. man.

(Kershaw, T., et al, 1980)

Health and Welfare Canada
Santé et Bien-être social Canada

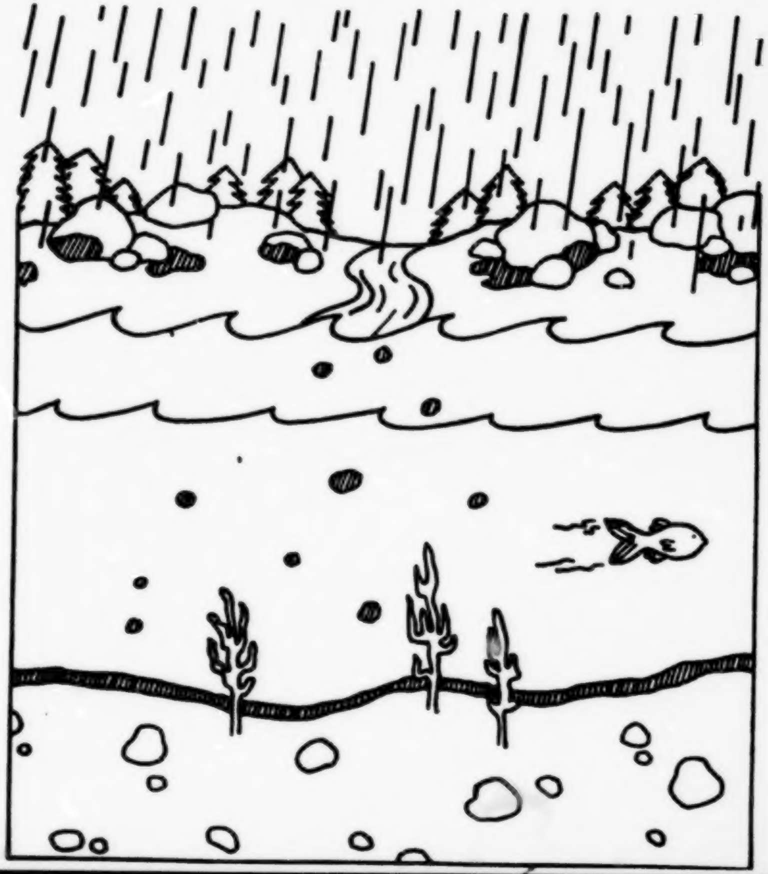
Environmental Contaminants Program

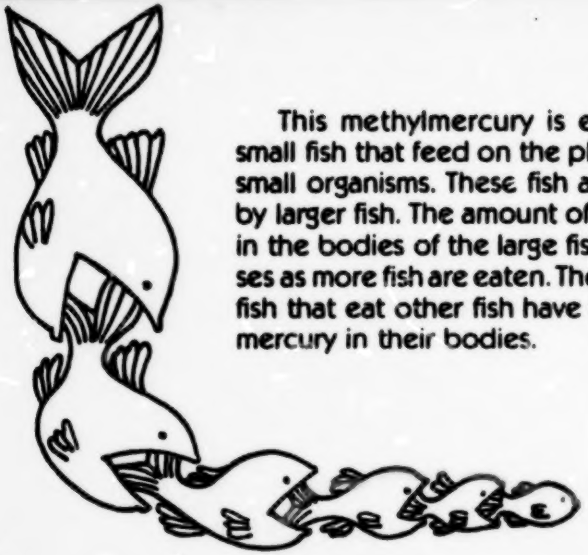
information about MERCURY

MEDICAL SERVICES BRANCH
MANITOBA REGION

Mercury has been in the earth since the beginning of time. During the passage of millions of years it has worked its way through the earth's surface appearing in rocks, soil, water, air and living organisms.

Naturally occurring mercury is broken down by the weather and washed into a river and lake system by the rain. This relatively harmless form of mercury is changed by tiny organisms in the water and sediments to the poisonous form called methylmercury. As a result there has been a lot of publicity and concern over this matter.

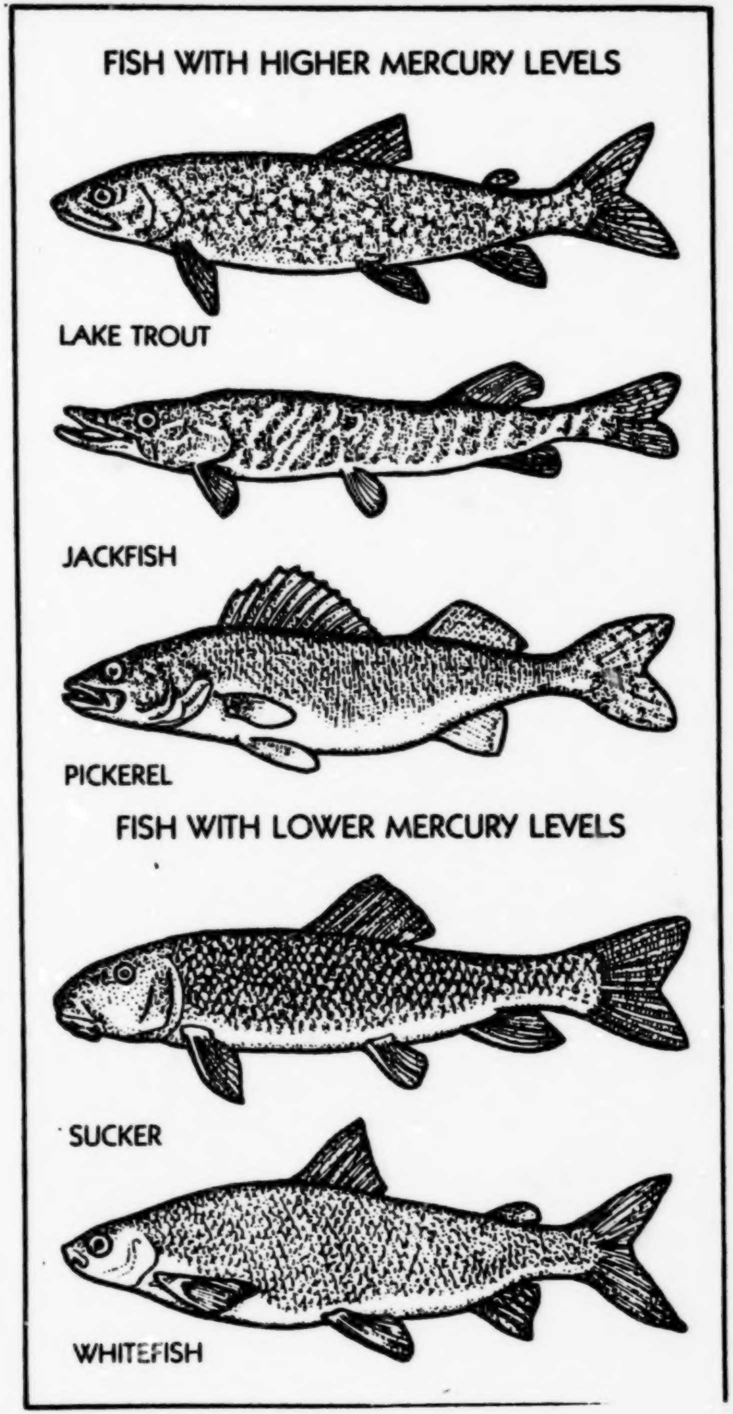




This methylmercury is eaten by small fish that feed on the plants and small organisms. These fish are eaten by larger fish. The amount of mercury in the bodies of the large fish increases as more fish are eaten. The kinds of fish that eat other fish have the most mercury in their bodies.

Fish with higher mercury levels are Pickerel, Jackfish, and Lake Trout. A person who eats a lot of these fish, increases the amount of mercury in his body to a level that might be harmful to his health.

Whitefish and Suckers have lower mercury levels. It is better to eat more of these fish and less of Pickerel, Jackfish, and Lake Trout.


FISH WITH HIGHER MERCURY LEVELS
LAKE TROUT
JACKFISH
PICKEREL
FISH WITH LOWER MERCURY LEVELS
SUCKER
WHITEFISH

In the mercury testing program particular attention has been paid to:

Fishermen and Guides
Because of their occupation, fishermen and fishing guides may eat more fish than most people and therefore may have a higher than average mercury level.

Pregnant Women
Although no harm has occurred in Canada, it is advisable to take a hair sample from women early in their pregnancy to determine their pattern of exposure to methylmercury and reduce their mercury levels if necessary.

Heads of Households
It has been found that the level of mercury in these persons usually represents the mercury levels in the other members of the household.

THE SIGNIFICANCE OF MERCURY LEVELS

The results of mercury levels are always given in terms of the blood level. If blood is sampled the result is recorded in parts per billion (ppb). If hair is sampled, it is tested at several points on the pieces of hair, and the highest result taken and converted into an equivalent blood level. This means that results from both hair and blood can be recorded together.

Most of the results in Manitoba are between 0 and 20 ppb, which is the normal range and would show no ill effects on health.

Those people with 20 to 100 ppb. have a higher than normal level, but no one in Canada with such levels has shown any signs of illness due to mercury. However, we do advise the people in this group to eat less of fish that may have high mercury levels (Pickerel, Jackfish and Lake Trout). We advise persons to eat Suckers and Whitefish since these fish have lower mercury levels than the other fish mentioned.

Certainly, in Manitoba, no one has shown signs of illness due to mercury even if their blood level is over 100 ppb. However, to be on the safe side, Medical Services arranges for them to be examined by a doctor and they are advised about eating fish.

Also to be on the safe side, pregnant women should have hair or blood tested early in pregnancy. If their level is over 20 ppb, they should eat less of the fish that may have a high mercury level.

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Published by authority of the
Minister of National Health and Welfare
Medical Services Branch — Manitoba Region
Date: April 1981

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